Appendix A: Management Unit Status Profiles

Nooksack River Management Unit Status Profile

Component Stocks

North/Middle Fork Nooksack early chinook
South Fork Nooksack early chinook

Geographic description

The Nooksack River natural chinook management unit is comprised of two early-returning, native chinook stocks that are genetically distinct, geographically separated, and exhibit slightly different migratory and spawning timing. They have been combined into a management unit because their similar migration timing through the fishing areas in the Nooksack River, below the confluence with the South Fork, and Bellingham and Samish Bays.

The North and Middle Forks drain high altitude, glacier-fed streams. Early-timed chinook spawn in the North Fork and Middle Fork from the confluence of the South Fork (RM 36.6) up to Nooksack Falls at RM 65, and in the Middle Fork downstream of the diversion dam, located at RM 7.2. Spawning also occurs in numerous tributaries including Deadhorse, Boyd, Glacier, Thompson, Cornell, Canyon, Boulder, Maple, Kendall, Racehorse, and Canyon Lake creeks. A hatchery-based egg bank and restoration program has operated at the Kendall Creek facility since 1981. Since then up to 2.3 million fingerlings, 142,458 unfed fry and 348,000 yearlings have been released annually into the North Fork, or at various acclimation sites. The yearling release program was discontinued after the 1996 brood because returns showed that survival rates were lower than those of fed fry releases. Since 2001, fingerlings have been released into the Middle Fork, in anticipation of removal of a blocking diversion dam. Beginning in 2003, the Kendall Creek program releases were downsized due to habitat capacity and straying concerns.

The South Fork drains a lower-elevation watershed that is fed primarily by snowmelt and rainfall, not by glaciers. Consequently, river discharges are relatively lower and temperatures relatively higher than the North and Middle forks during mid to late summer and early fall. Some South Fork tributaries have temperature regimes more similar to those in the North and Middle Forks during the late summer and early fall. A hatchery-based egg bank and restoration program operated at the Lummi Skookum Creek facility in brood years 1980 – 1993, but was discontinued when the returns to the hatchery ladder did not occur in significant numbers, and the capture of wild broodstock was not considered appropriate at such low abundances.

Life History Traits

Nooksack early chinook enter the lower Nooksack River from March through July, and migrate upstream over a 30 – 40 day period to holding areas. In the North / Middle Fork spawning occurs in the upper reaches from mid-July through late September, peaking in August. Spawning is currently concentrated in the North Fork, from RM 44 to RM 64, but may not represent the historical spawning distribution. The current distribution may be influenced by station and off-station release locations. Early chinook spawn in the South Fork from its confluence with the North Fork to a cascade at RM 30.4, and in Hutchinson, Skookum, Deer and Plumbago creeks. In the mainstem South Fork spawning is currently concentrated between RM 8 and RM 21. Hutchinson Creek has had the majority of the tributary spawning in recent years. South Fork spawning begins in August, and peak spawning occurs two to three weeks later than in the North / Middle Fork.

The North/Middle Fork Restoration Program utilizes several release strategies from the Kendall Creek Hatchery. Thermal otolith marks are applied to each release group, so their survival and spawning distribution can be evaluated when the fish return as adults. Otolith analysis has shown that strays into the South Fork, while small relative to the total returns of cultured fish to the watershed, can make up to 46% of the early stocks returning to the South Fork.

The release strategy in the of the North/Middle Fork restoration program was changed in 2001 to reduce the on-station release from Kendall Hatchery, which had shown the highest stray rate into the South Fork, from 900,000 fingerlings in 1998 in a series of reductions to 150,000 fingerlings in 2003, the current release goal. At the same time the total off-station release was reduced from 1,700,000 fingerlings in 1999 to 400,000 fingerlings in the North Fork, 200,000 in the Middle Fork, and 50,000 remote site incubator fry in the North Fork in 2003.

Earlier analysis of scales collected from North Fork spawners showed that a large majority (91%) emigrated from freshwater at age-0(WDFW 1995 cited in Myers et al 1998). In contrast, a larger and highly variable (as much as 69 percent) proportion South Fork spawners emigrated as yearling smolts. A more thorough, recent review of the adult scale data collected from natural-origin spawners, for those years when at least 40 samples collected, determined that 29% and 38% of North/Middle and South Fork early chinook, respectively, migrated from the river as yearlings. The number of naturally-produced fingerling and yearling smolts produced by the North / Middle and South forks has not been quantified.

Available information on the age composition of adults returning to the North/Middle forks and the South Fork is presented in Table 1, and indicate a predominance of age-4 returns. Age-5 proportions of these magnitudes are also observed among other Puget Sound spring chinook stocks, e.g. the Suiattle River and White River. Low sample sizes as a result of difficulties in recovering carcasses on the spawning ground require caution in the interpretation of this data.

Table 1. Estimates of the age composition of returning adult early chinook in the North / Middle and South Forks of the Nooksack River.

	Age 2	Age 3	Age 4	Age 5
North/Middle Fork NOR	1%	16%	73%	10%
South Fork NOR	0%	12%	72%	16%

Status

The current status of the Nooksack early chinook stocks is critical. The geometric mean number of natural-origin spawners in the North / Middle Fork, for 1998 – 2002, was 124, though NOR escapement has increased slightly in recent years from very low levels in the late 1990s (Table 2). The number of native, natural-origin spawners in the South Fork remains low, but is also apparently stable. The geometric mean NOR escapement in South Fork, for 1998 – 2002, was 224.

Table 2. Natural-origin escapement of early chinook to the North / Middle Forks and South Fork of the Nooksack River.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
No/Mid Fork	335	8	171	209	74	37	85	160	264	224
South Fork	235	118	290	203	180	157	166	284	267	289

Total natural spawning escapement has been substantially higher, due to returns from the Kendall Creek Hatchery supplementation program, which is considered essential to the protection and recovery of the North / Middle Fork population. In the North / Middle Fork, escapement has increased markedly since 1998, and exceeded 3,700 in 2002. The number of natural spawners in the South Fork has also increased, and reached 625 in 2002 (Table 3).

Table 3. The total number of natural early chinook spawners (i.e., hatchery- and natural-origin) in the North / Middle and South Forks of the Nooksack River. North / Middle Fork estimates exclude hatchery turnbacks.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
No Mid Fk	445	45	224	537	574	370	823	1242	2185	3741
South Fk	235	118	290	203	180	157	290	373	420	625

Survey effort has increased to better estimate the abundance and distribution of spawners throughout the Nooksack Basin, but turbidity due to the glacial origin of the North and Middle Forks hampers efforts to enumerate live fish or redds.

North/Middle Fork escapement in the last three years has been more than three times the average for the preceding five-year period (1992-96), while South Fork populations escapement has been stable at about 200 for the last five years. The recent increase in escapement to the North/Middle Fork (Table 4, Figure 1) is attributable in large part to the increase in releases from the Kendall Creek supplementation program, although earlier increases might be related to the reduction of Canadian harvest in the late 1990s. Recruits per natural-origin spawner in the North and Middle Forks have consistently remained below one recruit per pair of spawners. Preliminary estimates of the number of natural origin spawners in the North/Middle Forks, as determined from otolith studies, indicate that the return rate of natural origin spawners for brood years 1992 through 1995 ranged from 0.08 to 0.59 per spawner (Table 5), well below the replacement rate. The large and increasing number of hatchery-origin fish escaping to the North and Middle Forks suggests that harvest in the southern U.S. is not impeding the rebuilding of the abundance of natural origin spawners. The failure of the NORs to show a substantial increase in abundance similar to that of hatchery-origin fish, during the restricted fisheries in the late 1990s, suggests limitations in the ability of existing habitat conditions to support substantial productivity from the increased spawner abundance.

Table 4: Origin of Spawners in the North/Middle Forks of the Nooksack River (Co-Manager unpublished data).

Return Year	Natural Origin	Cultured Origin	Hatchery Turnbacks	Total
1995	171	53		228
1996	209	328		537
1997	74	500		574
1998	37	333		370
1999	85	738		823
2000	160	1082	891	2133
2001	264	1921	4802	6987
2002	224	3517	3731	7472

Figure 1. Natural-origin and total natural escapement to the North / Middle Fork of the Nooksack River, and Kendall Creek Hatchery releases three years prior.

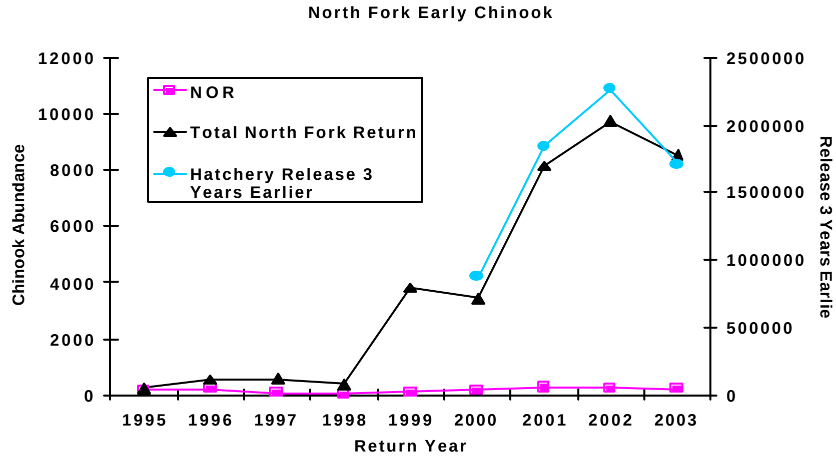


Table 5. Natural origin return per spawner rates for early chinook in the North/Middle Fork of the Nooksack River (Co-Manager unpublished data).

Brood year	Natural spawners	Total age 3 - 6 Returns	Return per Spawner
1992	493	185	0.38
1993	445	76	0.17
1994	45	25	0.56
1995	224	17	0.08
1996	533	247	0.46
1997	574	339	0.59
1998	370	103	0.36
1999*	823	149	0.18

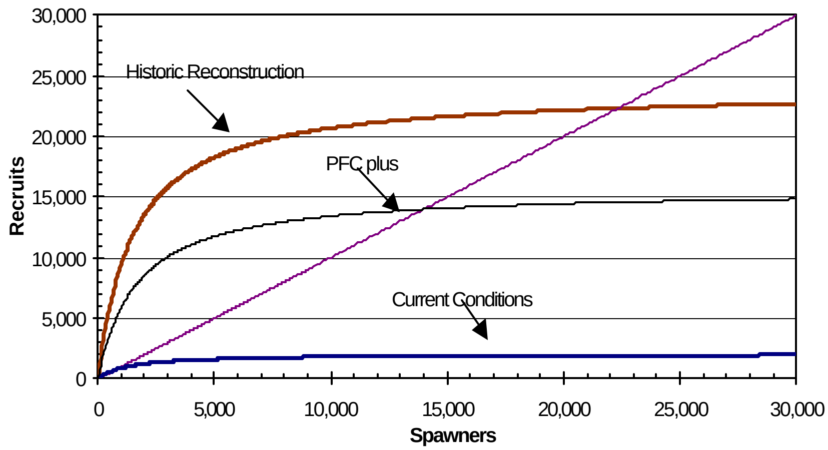
* age 3 and 4 returns only

While there is high variability in the relationship between natural-origin spawners and subsequent returns per spawner for the North / Middle Fork population, and statistical relationship is not significant, the data suggest that the recruitment rate is lower at higher spawner abundance. With the significant increase in natural spawners in recent years, the next four years will provide a clearer picture of the relationship between the number of spawners in the wild and the subsequent recruitment.

The Ecosystem Diagnosis and Treatment (EDT) methodology has produced habitat-based estimates of the productivity and abundance of the Nooksack early populations, under current, historical, and recovered (i.e. 'properly functioning' as identified by the NMFS in the FEMAT process) habitat conditions.

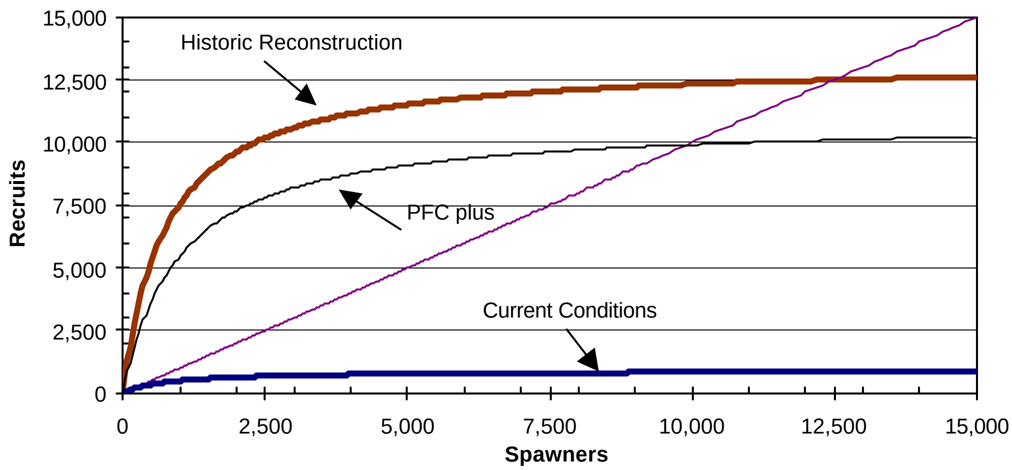
The EDT results for the North/Middle Forks under current conditions estimate capacity at 2,059 adults, equilibrium (i.e. replacement) abundance at 760, and productivity 1.6 adult recruits per spawner, without consideration of fisheries mortality. These results largely agree, but suggest slightly higher productivity than the spawner –recruit relationship derived directly from NOR escapements (Table 4). The EDT analysis indicates that productivity under recovered habitat conditions would be much greater (Figure 2).

Figure 2. Spawner-recruit relationships under current, recovered, and historical habitat conditions in the North / Middle Fork of the Nooksack River, as estimated by EDT analysis.



A similar analysis of the current productivity in the South Fork indicates adult capacity of 885, equilibrium (i.e., replacement) abundance of 80,and a return of 1.1 recruits per spawner. Productivity under recovered conditions would be far in excess of the current level. (Figure 3)

Figure 3. The spawner – recruit functions for South Fork Nooksack early chinook under current, recovered, and historic habitat conditions, as estimated by the EDT method.



The status of the South Fork stock is more difficult to determine in the absence of a reliable brood year return per spawner. The comparison of South Fork early escapement to the early escapement four years later suggest an average spawner replacement rate of 1.21 (Table 6). With the advent of otolith marks for each release strategy in the Kendall Creek Hatchery Program, the North/Middle Fork stock has been identified in the early chinook spawners in the South Fork. Because the 1991 release was the first to be otolith marked and pre-dated the substantial releases of cultured fish in the North and Middle Forks, it is assumed that the straying of North/Middle Fork chinook into the South Fork was low prior to 1995.

Table 6. Origin and replacement rate of early chinook spawners in the South Fork Nooksack River

Brood Year	South Fk Stock (no mark)	North Fk Stock	Stray Other or Unknown	Total	NOR BY+4	Replacement Rate
1991	365			365	290	0.79
1992	103			103	203	1.97
1993	235			235	180	0.77
1994	118			118	157	1.33
1995	166	87	37	290	166	0..57
1996	284	74	14	373	284	1.40
1997	267	138	15	420	267	1.48
1998	289	289	44	625	289	1.84
1999	204	217	148	570	204	0.70
					Average	1.21

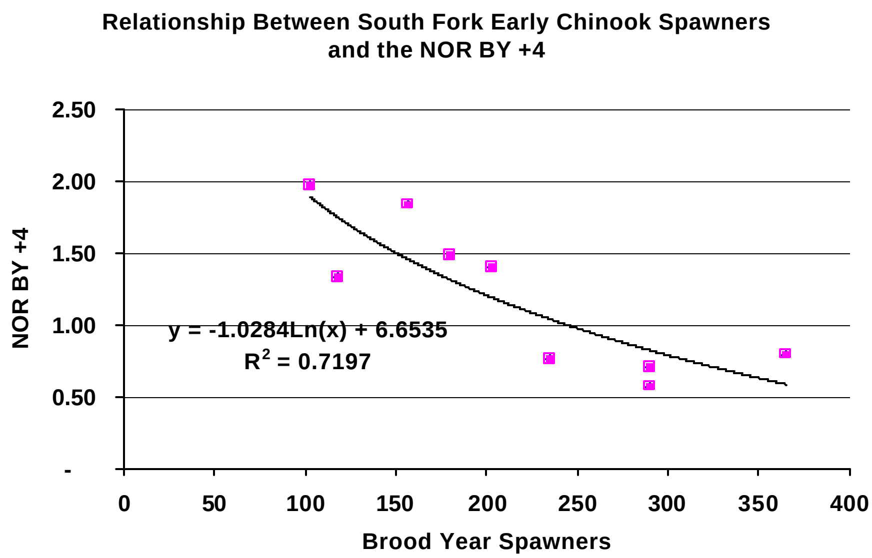
Recent information indicates that as much as 46% of the early chinook spawners in the South Fork have been strays from the Kendall Creek Hatchery program.

Table 7. Estimates of the contributions the native South Fork stock to natural spawning in the South Fork of the Nooksack River, 1999 - 2003.

Return Year	Total Early Number	South Fork Stock	
		Number	Percent
1999	290	166	57%
2000	373	284	76%
2001	420	267	64%
2002	625	289	46%
2003	570	204	36%

The relationship between the number of early chinook spawners in the South Fork and the number of natural origin recruits to the spawning grounds 4 years after the brood year (Figure 4) strongly suggests that habitat conditions constrain productivity in the South Fork. This relationship assumes that the reproductive success of the North Fork and other strays is similar to that of the South Fork population, and that the unmarked fish represent only NORs returning to the South Fork, regardless of the origin of the stock.

Figure 4. The relationship between natural origin early chinook spawners in the South Fork and their replacement rate for spawners four years later.



Harvest distribution

Recoveries of coded-wire tagged North Fork early chinook indicate that a majority of the historic harvest mortality occurs outside of Washington waters, primarily in Georgia Strait and other net and recreational fisheries in British Columbia (Table 8). The principles of abundance-based management of chinook, which were agreed to in the re-negotiated Pacific Salmon Treaty Chinook Annex in 1999, did not constrain harvest of Nooksack early chinook in Georgia Strait, where they comprise less than one percent of the total catch. Conservation measures aimed at reducing spring chinook harvest in the Strait of Juan de Fuca and northern Puget Sound have been in place since the late 1980s. There have been no directed commercial fisheries in Bellingham Bay and the Nooksack River since the late 1970's. Incidental harvest in fisheries directed at fall chinook in Bellingham Bay and the lower Nooksack River was reduced in the late 1980s by severely reducing July fisheries. Since 1997, there has been a very limited subsistence fishery in the lower river in early July. Commercial fisheries in Bellingham Bay that target fall chinook have been delayed until August for tribal fishers, and mid-August for non-treaty fishers. After 1997, the release of summer fall chinook from the Kendall hatchery was moved down to the tidal portion of the river and then to the Maritime Heritage Hatchery on the eastern shore of Bellingham Bay, and then eliminated entirely. Fall chinook production at the Lummi Sea Ponds facility was reduced by about 50% to about 1.0 million fingerlings in 1995. This has shifted the emphasis of the terminal area fishery away from the Nooksack River to the Samish Bay and Lummi Bay areas and reduced the proportion of the tribal harvest taken in the Nooksack River.

Table 8. Average harvest distribution of Nookack early chinook, for management years indicated, as percent of total adult equivalent fishery mortality (CTC 2003).

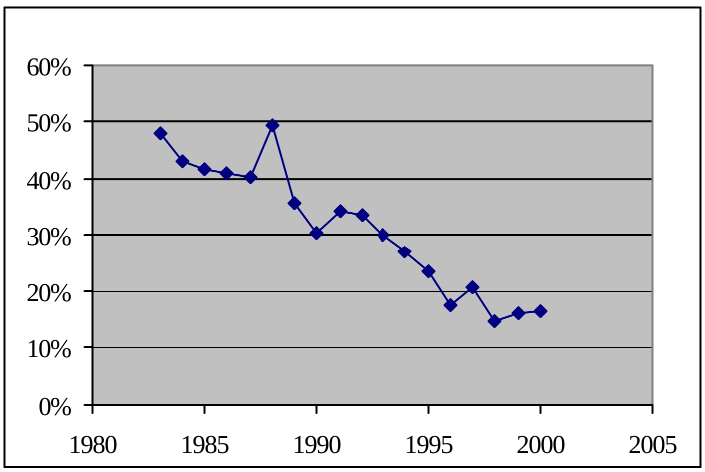
	Alaska	B.C.	Wa troll	PS net	Wa sport
1995-1999 yearlings	0.0%	67.4%	1.9%	6.4%	24.3%
1997-2001 fingerlings	21.5%	65.8%	3.0%	1.5%	8.2%

Coded-wire tag recoveries indicate that, in Washington waters, Nooksack early chinook have been caught in the Strait of Juan de Fuca troll fishery, recreational fisheries in southern and northern Puget Sound, and net fisheries (primarily in Areas 7 and 7A, Bellingham Bay, and the Nooksack River) in northern Puget Sound. The Kendall Creek facility currently releases only fingerling early chinook.

Exploitation rate trends:

The total annual fisheries exploitation rate for Nooksack early chinook, as estimated by post-season FRAM runs, has declined 59 percent, since the 1980s (Figure 1), from levels in excess of 40 percent in 1983 – 1988, to less than 20 percent in the last five years. Some uncertainty is associated with the absolute value of FRAM-based exploitation rates, but they are believed to accurately index the trend in rates. There are no current CWT data to enable a specific computation for the South Fork stock.

Figure 5. Total adult equivalent Exploitation rate of Nooksack early chinook for management years 1983 – 2000, estimated by post-season FRAM runs.



Management Objectives

Management objectives for Nooksack early chinook constrain harvest under co-manager jurisdiction so that it **will not impe de recovery**, while allowing for the exercise of treaty-reserved fishing rights and providing non-treaty fishing opportunity on harvestable salmon. The management objective will assure that natural-origin chinook, significantly in excess if MSY escapement levels under current conditions, escape to the spawning grounds to test existing habitat conditions to promote the recovery of the North / Middle and South Fork populations.

The upper management threshold for each Nooksack early population is set at 2,000 NOR spawners. The low abundance threshold for each population is 1,000 NOR spawners. For the next six years it is not expected that the abundance of natural origin spawners of either of the Nooksack early chinook stocks will exceed the low abundance threshold. Under this circumstance, fisheries that impact the escapement of these stocks will be shaped so a critical exploitation rate ceiling of 9% in southern US fisheries is not exceeded; the co-managers' intent is to constrain fisheries so that the projected SUS rate does not exceed 7% in more than once in the next six years.

The low abundance management threshold is currently under review and under current conditions may be significantly less than 1000 spawners. After reviewing the best available information the co-managers in consultation with NMFS may establish more appropriate low abundance management thresholds.

With 87% percent of the total annual harvest mortality occurring in Alaskan and Canadian fisheries (Table 8), the scope for total reducing fisheries impacts in Washington waters is limited. Net, troll, and recreational fisheries in Puget Sound have been shaped to minimize incidental chinook mortality to extent possible while maintaining fishing opportunity on other species such as sockeye and summer/fall chinook. The net fishery directed at Fraser River sockeye, in catch areas 7 and 7A in late July and August, has caught very few Nooksack early chinook.

Table 9. Estimates of the Origin of the Early Chinook Stocks Entering the Nooksack River.

Return Year	North Fk NOR	Total NF w/ Stray to SF	South Fk NOR	Total River Entry	SF+NF NOR	% NOR
1995	171	224	290	514	461	90%
1996	209	537	203	740	412	56%
1997	74	574	180	754	254	34%
1998	37	370	157	527	194	37%
1999	85	3820	166	3986	251	6%
2000	160	3426	284	3710	444	12%
2001	264	8146	267	8413	531	6%
2002	224	9723	289	10012	513	5%
2003	210	8519	204	8723	414	5%

There will be a limited ceremonial and subsistence harvest of Nooksack early chinook in the river, amounting to less than 10 natural origin spawners, and co-migrating cultured stock in excess of spawning requirements, as determined during preseason modeling. In addition, a limited tribal subsistence fishery, targeted at less than 20 natural origin spawners and co-migrating cultured stock in excess of spawning requirement, will occur in early July to meet minimum tribal requirements. These fisheries will occur from Slater Road crossing to the river mouth in the lower Nooksack, and from the Mosquito Lake road crossing down to the SR 9 bridge in the lower North Fork. The projected total harvest of early chinook by in-river tribal ceremonial and subsistence fisheries will be determined, during preseason planning, with reference to forecasted abundance of natural-origin and hatchery returns.

Fisheries in Bellingham Bay and the Nooksack River directed at fall chinook will not open prior to August 1. Subsequent fishing in the Nooksack River occurs in progressively more upstream zones as early chinook clear these areas. Thus the area extending two miles downstream of the confluence of the North and South Forks will not open prior to September 16.

Total exploitation rates projected by the FRAM model for the 2001 – 2003 management years were 18%, 15%, and 20%, respectively. The analysis supporting derivation of a rebuilding exploitation rate (RER) for the Nooksack MU is in progress. It is recognized that tag data do not exist to support a direct analysis of the productivity of the South Fork stock, and given its status, there is ample reason to exert conservative caution in planning fishing regimes.

The co-managers are evaluating the productivity, abundance and diversity of the early chinook runs that could be expected from the Nooksack watershed under properly functioning habitat conditions, as well as those that might have been expected to exist under historical conditions at Treaty time. The calculation of a normal exploitation rate has not be made but at the current escapement goal of 2000 natural origin spawners in each population, and an exploitation rate of 60%, a AEQ recruit abundance of 5,000 in each population would be anticipated. An ambitious and long-term effort to restore and protect habitat, working in concert with appropriate hatchery production and harvest management regimes, is essential to recovery.

Data gaps

Following are the highest priority needs for technical information necessary to understand stock productivity and refine harvest management objectives:

1) Improve estimates of population specific total escapement to the Nooksack basin, with emphasis on North/Middle and South Fork populations, including natural origin fish, and age data on these fish.
   a) Secure resources to read backlog of otoliths collected at the Kendall Creek hatchery to provide a complete evaluation of the contribution of the different release strategies.
   b) Improve the microsatellite DNA stock baselines of all chinook in the Nooksack Basin and conduct analyses to evaluate
      i) the NOR contribution of North/Middle Fork strays to the South Fork that can no longer be identified by otolith marks
      ii) the most appropriate break point to separate early and late chinook spawning in the South Fork
      iii) the relative success of chinook in the South Fork of the different populations as indicated by samples from the South Fork Smolt Trap
      iv) the relative success of North/Middle Fork spawners as indicated by samples collected at the Hovander smolt trap after eliminating the supplementation production identifiable by external mark (Calcein flourescense or fin clip)
   c) Develop alternative spawning ground population estimates that will allow:
      i) Update pre-spawning migration behavior through radio tags or DIDSON technology.
      ii) Increase recovery of carcasses on the spawning ground to improve estimates of the NOR age structure, yearling/sub-yearling contributions, and population composition.

2) Investigate rearing conditions in the river and the estuary and near shore areas to assist in the development of habitat restoration and protection actions.
3) Improve estimates of stock specific natural early chinook smolt outmigration from the North/Middle and South Fork populations and late timed chinook.
4) Develop stock/recruit functions, or other estimates of freshwater survival data to monitor the productivity of the two populations and late timed chinook.
5) Collect information to determine whether the current SUS fishing regime, or the hatchery supplementation program, are exerting deleterious selective effects on the size, sex, or age structure of spawners.

Skagit River Management Unit Status Profiles

Component Stocks

Summer/fall chinook management unit
- Lower Sauk River (summer)
- Upper Skagit River mainstem and tributaries (summer)
- Lower Skagit River mainstem and tributaries (fall)

Spring chinook management unit
- Upper Sauk River
- Suiattle River
- Upper Cascade River

Geographic description

There are two wild chinook management units originating in the Skagit River system - spring and summer/fall chinook. . The co-managers (WDFW and WWIT 1994) identified three spring and three summer/fall populations. The Puget Sound TRT concurred with this delineation in their assessment historical population structure (Currens et al. in prep. 2003).

Summer/fall management unit

The three populations tentatively identified within the summer/fall management unit are: Upper Skagit summers, Lower Sauk summers, and Lower Skagit falls. Upper Skagit summer chinook spawn in the mainstem and certain tributaries (excluding the upper Cascade River), from above the confluence of the Sauk River to Newhalem. Spawning also occurs in Diobsud, Bacon, Falls, Goodell, Illabot, and Clark creeks. Gorge Dam, a hydroelectric facility operated by Seattle City Light, prevents access above river mile (RM) 96, but historical spawning in the high-gradient channel above this point is believed to have been very limited. The lower Sauk summer stock spawns primarily from the mouth of the Sauk to RM 21 - separate from the upper Sauk spring spawning areas above RM 32. The lower mainstem fall stock spawns downsteam of the mouth of the Sauk River, and in the larger tributaries, including Hansen, Alder, Grandy, Jackman, Jones, Nookachamps, Sorenson, Day, and Finney creeks.

Skagit summer/fall stocks are not currently supplemented to a significant extent by hatchery production. A PSC indicator stock program collects summer broodstock (about 40 spawning pairs per year) from the upper river. Eggs and juveniles are reared at the Marblemount Hatchery. The objective of the program is to release 200,000 coded-wire tagged fingerlings for monitoring catch distribution and harvest exploitation rate. Summer chinook fingerlings are acclimated in the Countyline Ponds before they are released. Development of a lower river fall indicator stock was initiated in 1999, with similar production objectives. Production programs for fisheries enhancement of Skagit summer/fall chinook, and plants of fall chinook fingerlings into the Skagit system from the Samish Hatchery have been discontinued.

Spring management unit

The Skagit spring management unit includes stocks originating in the upper Sauk, the Suiattle, and upper Cascade rivers. The upper Sauk stock spawns in the mainstem, primarily above the town of Darrington up to RM 40, the Whitechuck River, and tributary streams. The Suiattle stock spawns in several tributaries including Buck, Downey, Sulphur, Tenas, Lime, Circle, Straight, and Big creeks. Cascade springs spawn in the mainstem above RM 19, and are thus spatially separated from the lower Cascade summer chinook. Spring chinook reared from Suiattle River broodstock are released from the Skagit Hatchery. Annual releases averaged 112,000 yearlings for the period 1982 – 1991 (WDF et al. 1993). Since then, about 250,000 subyearlings have also been released each year. All spring chinook releases are coded-wire tagged.

Life History Traits

The upper mainstem and lower Sauk River and summer stocks spawn from September through early October. Operational constraints imposed by the Federal Energy Regulatory Commission on the Skagit Hydroelectric Project's operation have, to some extent, mitigated the effects of flow fluctuations on spawning and rearing in the upper mainstem, and reduced the impacts of high flood flows by storing runoff from the upper basin. The lower river fall stock enters the river and spawns later than the summer stocks; spawning peaks in October. Age of spawning is primarily 4 years, with significant Age 3 and Age 5 fish. Most summer/fall chinook smolts emigrate from the river as subyearlings, though considerable variability has been observed in the timing of downstream migration and residence in the estuary, prior to entry into marine waters (Hayman et al. 1996).

Spring chinook begin entering freshwater in April, and spawn from late July through early September. Adult spring chinook returning to the Suiattle River are predominantly age-4 and age-5 (WDF et al. 1993 and WDFW 1995 cited in Myers et al. 1998). Glacial turbidity from the Siuattle River and Whitechuck River limit egg survival in the lower Sauk River. Analysis of scales collected from adults on the spawning grounds indicates that the proportion of spawners that outmigrated as yearlings ranged from 20% to 85% in the Suiattle, 35% to 45% in the Upper Sauk, and 10% to 90% in the Upper Cascade system.

Status

Stocks that comprise the summer/fall management unit are depressed. Annual spawning escapement has increased in the last five years (Table 1), but approached the critical threshold of 4,800 in 1997 and 1999. The geometric mean of the last five years' escapement was 12,690, an increase from the geometric mean of 1992-1996, 7,537 (Myers et al. 1998). Recent assessment of freshwater productivity for summer/fall chinook suggests that the current MSY escapement is about 14,500 (see below).

Table 1. Spawning escapement of Skagit River chinook, 1992- 2002.

	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Sauk sum	469	205	100	263	1103	295	460	295	576	1103	910
U Skagsum	5548	4654	4565	5948	7989	4168	11761	3586	13092	10084	13815
L Skag fall	1331	942	884	866	1521	409	2388	1043	3262	2606	4866
S/F MU	7348	5801	5549	7077	10613	4872	14609	4924	16930	13793	19591
Cascade sp	205	168	173	226	208	308	323	83	273	625	340
Siuattle sp	201	292	167	440	435	428	473	208	360	688	265
Sauk sp	580	323	130	190	408	305	290	180	388	543	460
Sprg MU	986	783	470	856	1051	1041	1086	471	1021	1856	1065

Spawning escapement for the spring unit has been consistently below 2,000, but has, with the exception of 1994 and 1999, been above the critical abundance threshold of 576. The geometric mean of escapement in 1998 – 2002 was 1,006.

Harvest distribution

Coded-wire tag recovery data for PSC indicator stocks provide a description of the harvest distribution of Skagit chinook, and contrast the differences between summer / fall and spring stocks. Yearling and fingerling releases from Marblemount Hatchery describe the distribution of spring chinook. The Samish Hatchery fall fingerling releases are believed to provide an accurate surrogate for describing the distribution of Skagit summer / fall chinook. Local summer and fall indicator stocks are being developed. Approximately 33 percent of the mortality of summer / fall chinook has occurred in fisheries in British Columbia and Alaska (i.e. outside the jurisdiction of the Washington co-managers). Twelve percent of summer / fall chinook are caught in Washington ocean fisheries. Puget Sound net fisheries and Washington sport fisheries accounted for 54 percent and 11 percent, respectively, of total summer / fall fishing mortality (Table 2). The harvest distribution of yearling and fingerling spring chinook differ, with about 51 and 75 percent of mortality occurring in northern fisheries, respectively. Puget Sound net fisheries account for 4 percent. Washington recreational fisheries account for 43 percent of yearling mortality, and 20 percent of fingerling mortality.

Table 2. Average harvest distribution of Skagit River chinook, for management years 1997 – 2001, as percent of total adult equivalent fishery mortality (CTC 2003 in press)

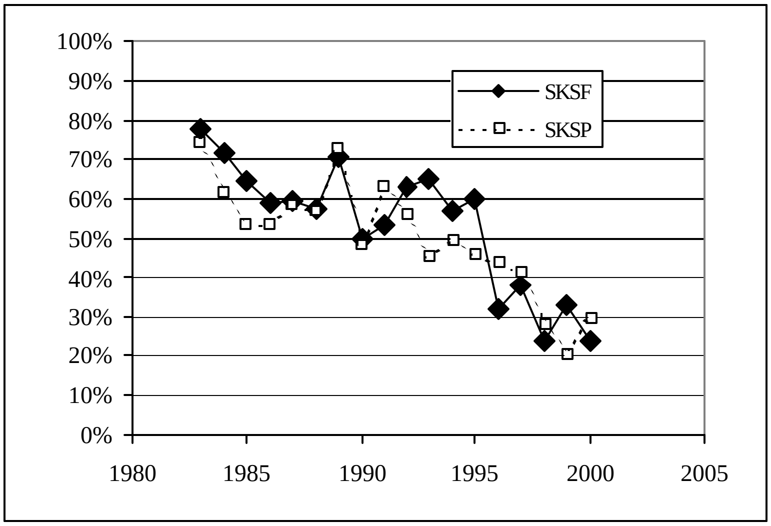
	Alaska	B.C.	Wash. Ocean	Puget Sound Net	Washington sport
Summer Fall	2.6%	30.5%	1.9%	54.1%	11.0%
Spring yrlng	1.1%	50.2%	1.8%	4.2%	42.7%
Spring fing	7.6%	67.6%	0.5%	3.8%	20.5%

Coded wire-tagged Skagit summer and fall indicator stocks, reared from indigenous broodstock at the Marblemount Hatchery, are now being released, and will allow more accurate estimation of harvest distribution and exploitation rates.

Exploitation rate trend:

Annual (management year) exploitation rates for Skagit summer/falls, as estimated by post-season FRAM runs, , have fallen 60 percent, from levels in excess of 60 percent in 1983 – 1987, to an average of 27 percent in 1998 - 2000. Over the same period, exploitation rates for spring chinook have fallen 57 percent, from similar historical levels to a recent average of 26 percent (Figure 1).

Figure 1. Total AEQ fisheries exploitation rate of Skagit summer / fall and spring chinook, estimated from post-season FRAM runs for management years 1983 – 2000.



Management Objectives

Derivation of Upper Management Thresholds

The Puget Sound chinook Evolutionarily Significant Unit (ESU) was listed as "threatened" under the Endangered Species Act in 1999, reflecting the overall poor abundance of the ESU (Myers *et al*. 1998). While the overall abundance of the ESU is poor, and fisheries have been significantly reduced as a result (Puget Sound Indian Tribes and Wash. Dept. Fish and Wildlife 2003), there may exist, from time to time, management units within the ESU that have relatively high abundance, which could support additional harvests. In order to access these harvestable fish, the abundance level that can support additional harvests must first be quantified for each management unit

In the harvest management component of the Puget Sound Comprehensive Chinook Management Plan ("Comprehensive Chinook"), this threshold for harvestable abundance (hereafter, "upper management threshold") is expressed as a spawning escapement level. Under this plan, a management unit has harvestable abundance if, after accounting for expected Alaskan and Canadian catches, and incidental, test, and tribal ceremonial and subsistence catches in southern U.S. fisheries, the spawning escapement is expected to exceed this level, and the unit's projected exploitation rate is expected to be less than its exploitation rate (ER) ceiling. In such cases, additional fisheries, including directed fisheries (fisheries in which this unit comprises the majority of the catch), may be implemented until either the ER ceiling is met, or the expected escapement equals the management threshold (Puget Sound Indian Tribes and Wash. Dept. Fish and Wildlife 2003).

Under the court-ordered Puget Sound Salmon Management Plan, the default threshold that defines harvestable surplus is the level that provides maximum sustained harvest. This objective can, however, be modified by co-manager agreement. For the Skagit summer/fall and spring chinook management units, recognizing the inherent variability in forecasting and recruitment, we define the management threshold as the escapement level that, within the framework of Comprehensive Chinook, is most likely to maximize the long-term catch of that unit. This paper describes the methods used to calculate those thresholds for both Skagit chinook management units.

Methods

Given this definition, the upper management threshold can be calculated analytically. To do this analysis, I wrote a QuickBasic program (CkUBPAge.BAS) (Appendix I) that simulates recruitment, catches, and escapement over a selected period of years, under conditions of uncertainty and error in management, and environmental variation. Because each Skagit chinook management unit is believed to be composed of three separate populations, I wrote this program to simulate up to six populations, each of which can have different productivity and capacity. To mimic current management, the harvest rate is applied on a calendar year basis; thus, each age that matures in a given year experiences the same harvest rate, but each age within a cohort can be harvested at a different rate.

Before doing the modeling, however, it was necessary to resolve three input and modeling questions:

Do we use spawner-recruit parameters that apply to current habitat conditions, or to properly functioning conditions (PFC)?

Because we lack agreed recruitment values for the separate Skagit chinook populations, I used spawner-recruit parameters that had been derived from a habitat-based method, Ecosystem Diagnosis and Treatment (EDT) (Lichatowich *et al.* 1995; Mobrand Biometrics 1999), to get the population-specific spawner-recruit parameters. But because EDT gave Beverton-Holt spawner-recruit parameters under historic conditions and PFC, as well as current conditions, we had a choice to make: which set of parameters should we use for this modeling?

The co-manager policy decision was to use current habitat conditions. The ER ceilings were calculated under assumed current survival rates, so it seemed consistent to assume current conditions when setting the management thresholds. In response to questions about whether this assumption would be responsive to any improvements in habitat, it was noted that these thresholds will be re-evaluated after 5 years, and also that harvest rates would be limited to the current ER ceiling, so if productivity did improve, constraining harvests to the current ER ceiling would allow for escapements to increase above the management threshold. Analyses for Snohomish chinook indicated that, while the calculated MSY escapement under current conditions (approximately 3,000) has been exceeded only 32% of the time in past years, if habitat improved to PFC, and the ER ceiling calculated under current conditions (24%) remained in place, the new MSY escapement (approximately 6,000) would be exceeded 95% of the time, even though the MSY escapement doubled (C. Kraemer, WDFW, *pers. comm.*).

Which point of instability estimates would be used for the summer/fall populations?

For Skagit summer/fall chinook, two sets of point of instability estimates were available: a set derived in 1999 (J. Scott, WDFW, *pers. comm.*), which has been used by NOAA Fisheries for their assessments, and 5% of the EDT-derived historic capacity (5% of capacity is a rule-of-thumb point of instability estimate discussed in Peterman 1977).

Empirical observations indicated that the EDT-derived estimates were too high. In 5 of the last 10 years, Lower Skagit and Lower Sauk escapements were both below the EDT-derived numbers, and in each case, the recruits/spawner rate was well above 1.0 (my program assumes that recruits/spawner averages 1.0 for escapments below the point of instability). During that same time, we did have one Lower Sauk escapement that was also less than its 1999-estimated point of instability, and the recruits/spawner rate for that brood was also well above 1.0, which indicates that that number may also be an overestimate of the point of instability, but, lacking any alternatives, I used the set of estimates derived in 1999 as the points of instability for Skagit summer/falls (Table 1).

Because there were no alternative estimates from earlier years for Skagit springs, and the EDT-derived estimates were the only ones available, I used 5% of the EDT-derived historic capacity as the points of instability for Skagit springs (Table 1). There have been no observed escapements below this point for Suiattle springs, and one near that level for the Upper Cascade population; however, that was in 1999, and the returning brood has not yet fully recruited. For Upper Sauk springs, there have been three observations below its point of instability, two of which have fully recruited, and in both cases the recruits/spawner rate exceeded 1.0.

When modeling a regime that includes a directed fishery, should the denominator used in the calculation of the target ER be the predicted recruitment, or the actual recruitment?

When there is a directed fishery, I modeled the target harvest rate as the harvestable number divided by the recruitment (see Step 8c below). The question was whether the denominator in that calculation should be the predicted recruitment or the actual recruitment. I decided that using the predicted recruitment more accurately simulates our real-world management, in which harvestable numbers are calculated according to predictions; therefore, I used the predicted recruitment in the denominator of that equation.

With these modeling and input questions answered, the steps used to generate the upper management thresholds are as follows:

1. Set the initial inputs. Run-specific inputs are the range of management thresholds that will be tested, the number of runs for each management threshold (each of which starts with a different random number sequence), the number of years for each run, and the populations that will be modeled in the run. Management inputs are the management error distribution, the forecast error distribution, the distribution of freshwater peak flows and marine survival, and the management unit-specific ERs: the ceiling ER, the average ER under incidental fisheries only, the average ER when abundance is critical, the minimum possible ER, and the maximum possible ER. Population-specific inputs are the Beverton-Holt spawner-recruit parameters, point of instability (the escapement level below which the mean recruits/spawner is 1), cohort age composition, initial escapements, and initial recruitments for the ages that precede the recruitments that result from the initial escapements. These inputs are listed in Tables 1 to 5.

2. Set the management threshold.

3. Seed the random number generator

4. Begin each year of a run. Simulate environmental variation that year by multiplying a randomly-chosen freshwater survival factor (Table 4) by the exponential of a cyclically-generated marine survival factor (Table 5). The marine survival factor is of the form:

$$\text{Factor} = A * \sin((\text{Year} / c) + b - 1/c) + s_{sine} * e$$

Where A is half the amplitude of the sine curve; b is the starting point on the sine curve, in radians, in Year 1 of the run, with b set at the start of each run to vary randomly between -2p and 2p (i.e., the marine survival cycle can start in Year 1 of each run anywhere from the beginning of the down cycle to the beginning of the up cycle); c * 6 gives the approximate period of the cycle (e.g., c = 4 gives about a 24-year cycle); 1/c is an adjustment I needed to account for starting the run in Year 1, rather than Year 0; s_{sine} is the standard deviation of the spread around the sine curve; and e is a normally-distributed error variable with a mean of 0 and standard deviation = 1. A and c were calculated by fitting a sine curve by least squares to the natural logarithms of the 1980-1992 marine survival indices provided by Jim Scott (J. Scott, WDFW, *pers. comm.*) (Table 5; Fig. 1). s_{sine} is the standard deviation of those indices around that fitted curve.

5. From the spawning escapements that have been initially input or calculated through the program, and the environmental variation factor produced in Step 4, use the Beverton-Holt parameters to generate the population-specific recruitments that will result in 3 to 5 years, and distribute them by age according to the cohort age composition of the population.

6. Sum the age-specific and population-specific recruitments that apply to the current year to calculate the current year's true total recruitment.

7. Multiply the true recruitment by a randomly-chosen forecast error value (Table 2) to calculate the current year's forecasted total recruitment.

8. Using the forecast, generate the current year's target ER. Assume initially that the ER is the average ER under incidental fisheries. If:

   a) The resulting escapement would be less than the sum of the points of instability for all populations modeled, then the critical abundance ER becomes the target;

   b) Otherwise, if the resulting escapement would be less than the management threshold, then the average ER under incidental fisheries remains the target;

   c) Otherwise, the harvestable number is the lesser of the difference between the recruit forecast and the management threshold, and the recruit forecast multipled by the ER ceiling. The target ER becomes the harvestable number divided by the recruit forecast.

9. Divide the target ER by a randomly-chosen management error value (Table 3), to generate the actual ER. Constrain this ER so that it is between the minimum and maximum possible ERs (Table 1).

10. Multiply the actual ER by the true recruitment to generate the catch, and multiply each population-specific and age-specific component of the true recruitment by the complement of the actual ER to get the escapement by population.

11. Go to Step 4 and repeat for 40 years.

12. Increment the random number generator, go to Step 3, and repeat 1000 times.

13. Go to Step 2 and use a different management threshold. Continue until I've identified the management threshold that produces the highest mean catch. That level becomes the management threshold for the Skagit chinook unit being examined.

Results

In preliminary model runs, I tested the sensitivity of the model results to three inputs that are fairly arbitrary: the number of years per run; the number of runs (each started with a different random number seed) for each management threshold tested; and the starting random seed. The results were not affected by the number of runs (the minimum number I tested was 1000 runs) or by the random seed; however, the estimate of the summer/fall chinook management threshold that maximized long-term catch was sensitive to the number of years per run (more years/run gave higher management thresholds). This sensitivity occurred because, as modeled, when abundance drops below the point of instability, it tends to stay there. If this occurs in, e.g., year 20 of a 25-year run, the long-term average catch gets depressed for only 5 years, whereas catch can be depressed for 20 years if this occurs in year 20 of a 40-year run. So there's more of a penalty to falling below the point of instability in longer runs. Since it's more likely that abundance will drop below the point of instability when the management threshold is lower, the runs with more years should favor higher management thresholds.

So I had a subjective decision to make: what should be the number of years per run? I chose 40 years/run (Table 1), feeling that this provided a middle-ground on the penalty for letting abundance fall below a point of instability – more than a 25-year run, and less than a 100-year run (the lengths of the runs were also limited by the amount of time it took to run the program). A 40-year run is about 10 generations of chinook salmon, and approximately 2 marine survival cycles, which I felt provided a sufficient range of variability in the analysis.

Skagit summer/fall chinook:

The maximum mean modeled catch, 13,094, occurred at management thresholds of both 14,000 and 15,000 (Table 6). I therefore split the difference, thereby deriving a Skagit summer/fall chinook management threshold of 14,500. As explained above, I used 40-year runs to derive this threshold. If I had used 25-year runs (which is the time period that was used to establish the ceiling ERs), the maximum mean modeled catch would have occurred at a management threshold of 12,000. With 100-year runs, the maximum mean modeled catch would have occurred at a management threshold of 16,000.

Skagit spring chinook:

The maximum mean modeled catch, 1598, occurred at management thresholds of both 2000 and 2100 (Table 7). Splitting the difference would give a management threshold of 2050. However, while rounding the threshold to the nearest hundred is consistent with other Puget Sound chinook goals, rounding to the nearest ten isn't. So the choice was between 2000 and 2100, and, since the previous Skagit spring chinook goal had been rounded to the nearest thousand (3000), the co-managers agreed to use 2000 as the management threshold for Skagit spring chinook. For springs, the management threshold was not sensitive to the number of years/run; with both 25-year runs and 100-year runs, the management threshold would still have been 2000.

Discussion

It might be argued that there is not much difference between the average catches shown in Tables 6 and 7, and that a different management threshold might be selected with little effect on long-term catch. That may or may not be true (I didn't examine the degree of fluctuation between individual catch years). However, the intent of this exercise was to calculate an answer that had *a* *single solution* that would achieve previously-defined criteria, in order to avoid the conflicts that result from trying to agree on arbitrary buffers or numbers that "look good". In this case, the criterion was maximization of mean catch, no matter how small the difference in mean catch. And, while there was subjectivity involved in some of the inputs (e.g., years/run – see above), it was objective in that the analysis yielded a single solution.

The proposed management thresholds, 14,500 for summer/falls and 2,000 for springs, are considerably higher than the MSY escapement levels that would be calculated analytically, without consideration of management error and environmental variation, from the spawner-recruit parameters listed below. From the parameters listed below, using Ricker's (1975) formulae for computing MSY escapement levels in a Beverton-Holt function, the MSY escapement levels under current conditions would be 7,700 for summer/falls and 900 for springs. Thus, by accounting for observed levels of management error and bias (both the forecasts and the target exploitation rates have tended to be overestimates of the post-season numbers – see Tables 4 and ?), and environmental variation, and by assuming the incidental catch rates observed in recent years under the Comprehensive Chinook framework, the management thresholds that maximize long-term catch are approximately double the MSY escapement levels calculated from formulae that do not account for those factors.

For summer/falls, this management threshold of 14,500 is almost the same as the former spawning escapement goal, 14,900, that was set in 1977. It is somewhat surprising that the two numbers are so close, since the former goal was nothing more than the average escapement calculated for the years 1965-1976 (Ames and Phinney 1977), and no analysis of production relationships was involved in its calculation.

For Skagit springs, on the other hand, the management threshold of 2,000 is considerably lower than the former spawning escapement goal of 3,000, which was set in 1975. This former goal was also calculated only as the average of escapements from an earlier period of years (1959-1973 in this case), rounded to the nearest thousand (Management and Research Division 1975), and the fact that the currently-calculated threshold is significantly different is not a great surprise, especially given that the biologists who now do the spawning escapement estimates have expressed considerable skepticism about the accuracy of the escapement estimates from those earlier years (P. Castle, WDFW, *pers. comm.*). In addition, it has been noted (C. Kraemer, WDFW, *pers. comm.*) that, with exploitation rates on springs slashed by about 70% in recent years, it would be expected that there would be a significant increase in resulting run sizes if there is a lot of unused capacity in the system. The fact that run sizes have instead remained fairly stagnant probably indicates that recent escapement levels (the highest in recent years was about 1900) are not far under the system capacity. By this reasoning, therefore, using directed fisheries to crop off escapement, when the escapement is expected to exceed 2,000, would be unlikely to detract from future production.

In summary, the calculated upper management thresholds for Skagit chinook are:

Skagit summer/fall chinook:	14,500
Skagit spring chinook:	2,000

Table 3. Input values used to generate management thresholds for Skagit summer/fall and spring chinook. See Tables 4 to 6 and Appendix I for data sources.

Run-Specific Inputs:
Number of years/run: 40
Number of runs: 1,000
Initial random seed: -15,000
Increment between seeds: 1

Management and Environmental Inputs:
Forecast Error: (See Table 2)
Exploitation Rate Error: (See Table 3)

ER Inputs:	Summer/Fall Chinook	Spring Chinook
Ceiling ER	52%	42%
Mean ER Under Incidental Fisheries	34%	28%
Mean ER Under Critical Abundance	29%	25%
Minimum Possible ER	15%	6%
Maximum Possible ER	90%	90%

Distribution of Peak Flows: See Table 6
Marine Survival Parameters (see Table 7 for the historic indices):
A (half of amplitude): 0.53
Period: 24 years
c (period/6): 4
s_{sine}: 0.633
Maximum Deviation Factor from Spawner-Recruit Curve: 5.0
Minimum Deviation Factor from Spawner-Recruit Curve: 0.1

Population-Specific Inputs:

	Up Skagit Summers	Lo Skagit Falls	Lo Sauk Summers	Up Sauk Springs	Suiattle Springs	Up Casc Springs
Bev-Holt a	17,600	10,600	4,500	2,600	500	900
Slope at Origin	9.2	3.3	5.9	8.5	8.2	8.0
Point of Instability	967	251	200	210	40	80
% Age 3	25%	25%	25%	5%	5%	5%
% Age 4	60%	60%	60%	59%	59%	59%
% Age 5	15%	15%	15%	36%	36%	36%
Initial Escapement	9,600	2,300	610	350	430	330
Initial Recruitment	Calculated by age as Initial Escapement/(1-Incidental ER) * Age Comp					
Extinction Level	10	10	10	10	10	10

Table 4. Run size estimation error values used in the program to generate management thresholds for Skagit summer/fall and spring chinook. The in-season update (ISU) error was used, rather than the preseason forecast error, because directed fisheries (which would be conducted if the escapement is predicted to exceed the management threshold) would most likely be managed according to an in-season update.

	Year	ISU	Post-Season	Difference	% Error (ISU/Post - 1)
	1984	15838	16791	-953	-5.7%
	1985	23360	25444	-2084	-8.2%
	1986	18583	22500	-3917	-17.4%
	1987	17347	13542	3805	28.1%
	1988	18992	16229	2763	17.0%
	1989	21403	13568	7835	57.7%
	1990	16586	20615	-4029	-19.5%
	1991	17382	9707	7675	79.1%
	1992	17933	11855	6078	51.3%
	1993	15150	8255	6895	83.5%
Mean		18257	15851	2407	26.6%
Std Dev		2507	5597	4782	39.4%
SE Mean		793	1770	1512	12.5%

Table 5. Exploitation rate error values used in the program to generate management thresholds for Skagit summer/fall and spring chinook. The error values used in the program are the 1988-93 and 1997-2000 rates listed in the two right-hand columns, under "S/F Ck" and "Spr Ck". The 1997-2000 values were calculated from the validation (post-season) and FRAM ER Index (preseason) values shown in this table. The 1988-1993 error values were calculated by Gutmann (1998).

	Validation Run		FRAM ER Index		FRAM Preseason U		% Difference (PSF/Validation - 1)		
Year	S/F Ck	Spr Ck	S/F Ck	Spr Ck	S/F Ck	Spr Ck	S/F Ck	Spr Ck	Combined
1988	58%	59%					22.6%	8.1%	
1989	71%	75%					-10.1%	-17.7%	
1990	50%	50%					12.6%	-0.6%	
1991	53%	65%					-7.1%	-16.2%	
1992	63%	57%					-12.7%	-6.9%	
1993	65%	46%					-18.6%	20.8%	
1994	57%	51%							
1995	60%	47%							
1996	30%	45%							
1997	37%	42%	85.0%	80.6%	51.3%	47.3%	38.7%	12.5%	
1998	23%	30%	62.7%	53.6%	37.9%	31.4%	64.6%	4.7%	
1999	33%	23%	74.9%	74.4%	45.2%	43.6%	37.1%	89.6%	
2000	24%	32%	45.2%	39.4%	27.3%	23.1%	13.8%	-27.9%	
2001			62.8%	37.7%	37.9%	22.1%			
2002			40.7%	41.4%	24.6%	24.3%			
2003									
89-93 avg	60.4%	58.6%					-2.2%	-2.1%	-2.2%
97-02 avg	29.3%	31.8%	61.9%	54.5%	37.4%	31.9%	38.5%	19.7%	29.1%
all yrs avg							14.1%	6.6%	10.4%
Std Dev							27.0%	32.8%	29.5%
SE Mean							8.5%	10.4%	6.6%

Table 6. Freshwater flow survival values for Skagit chinook. The values used in the program to compute management thresholds are those in the column labeled "Ratio to Mean". "RI" is flood return interval. Survival rates were calculated from a relation between flood return interval and incubation survival, using survival vs. peak flow data provided by Seiler *et al.* (2002), and converting peak flow to a flood return interval (E. Beamer, Skagit System Cooperative, *pers. comm.*).

Date	Brood Year	Survival	Ratio to Mean	Peak Discharge	RI (yr)
December 26, 1972	1972	17.5%	1.15	53600	1.8
January 16, 1974	1973	16.0%	1.05	77600	4.3
December 21, 1974	1974	17.6%	1.15	51400	1.6
December 4, 1975	1975	6.2%	0.40	130000	30.9
January 19, 1977	1976	17.6%	1.15	52800	1.7
December 3, 1977	1977	16.9%	1.11	65600	2.8
November 8, 1978	1978	18.0%	1.18	40300	1.1
December 19, 1979	1979	10.6%	0.69	112000	15.7
December 27, 1980	1980	10.2%	0.66	114000	17.0
February 16, 1982	1981	17.5%	1.14	55800	1.9
December 4, 1982	1982	16.5%	1.08	71600	3.5
January 5, 1984	1983	14.8%	0.97	88200	6.5
January 0, 1900	1984	18.0%	1.18		1.0
January 19, 1986	1985	16.4%	1.07	72800	3.6
November 24, 1986	1986	16.6%	1.08	70700	3.4
December 10, 1987	1987	18.2%	1.19	32100	0.8
October 17, 1988	1988	17.4%	1.14	56700	2.0
December 5, 1989	1989	13.4%	0.88	97800	9.2
November 25, 1990	1990	1.5%	0.10	152000	70.3
February 1, 1992	1991	18.0%	1.18	40100	1.1
January 26, 1993	1992	18.3%	1.19	27600	0.7
December 11, 1993	1993	18.2%	1.19	32100	0.8
December 28, 1994	1994	17.3%	1.13	58600	2.1
November 30, 1995	1995	3.5%	0.23	141000	46.6
January 20, 1997	1996	17.7%	1.15	50800	1.6
October 5, 1997	1997	17.0%	1.11	64800	2.7
December 14, 1998	1998	17.3%	1.13	58200	2.1
November 13, 1999	1999	16.1%	1.05	76000	4.1
October 21, 2000	2000	18.3%	1.19	26700	0.6
January 8, 2002	2001	16.5%	1.08	71900	3.5
Mean		15.3%	1.000	70441	8.2
Std Dev		4.4%	0.290	33040	
SE Mean		0.81%	0.053	6135	

Table 7. Values used to fit a sine curve to the natural logarithm of the marine survival index for Skagit summer/fall chinook. Period of cycle is approximately 24 years.

Brood Year	Marine S Index	ln(index)	aSin((Yr+b)/c)	Deviation	Dev-squared
80	0.755	-0.2810	0.52832	-0.8094	0.655059
81	4.313	1.4616	0.501463	0.9602	0.921928
82	1.232	0.2086	0.443427	-0.2348	0.055126
83	1.281	0.2476	0.357822	-0.1102	0.01214
84	1.783	0.5783	0.249969	0.3283	0.1078
85	0.413	-0.8843	0.126574	-1.0109	1.021881
86	2.352	0.8553	-0.00469	0.8600	0.739526
87	0.739	-0.3025	-0.13566	-0.1668	0.02782
88	0.775	-0.2549	-0.2582	0.0033	1.1E-05
89	0.801	-0.2219	-0.36469	0.1428	0.02039
90	1.66	0.5068	-0.4485	0.9553	0.912626
91	0.293	-1.2276	-0.50442	-0.7232	0.522962
92	0.374	-0.9835	-0.52898	-0.4545	0.206585
				SSE	5.20385
Mean	1.290077	-0.02288		MSE	0.400
Median	0.801	-0.22189		RMSE	0.63269

a = 0.53
b = 2
c = 4

Figure 2. The best fit sine-curve to Skagit summer/fall chinook marine survival indices for brood years 1980-1992. The period of the curve is about 24 years.

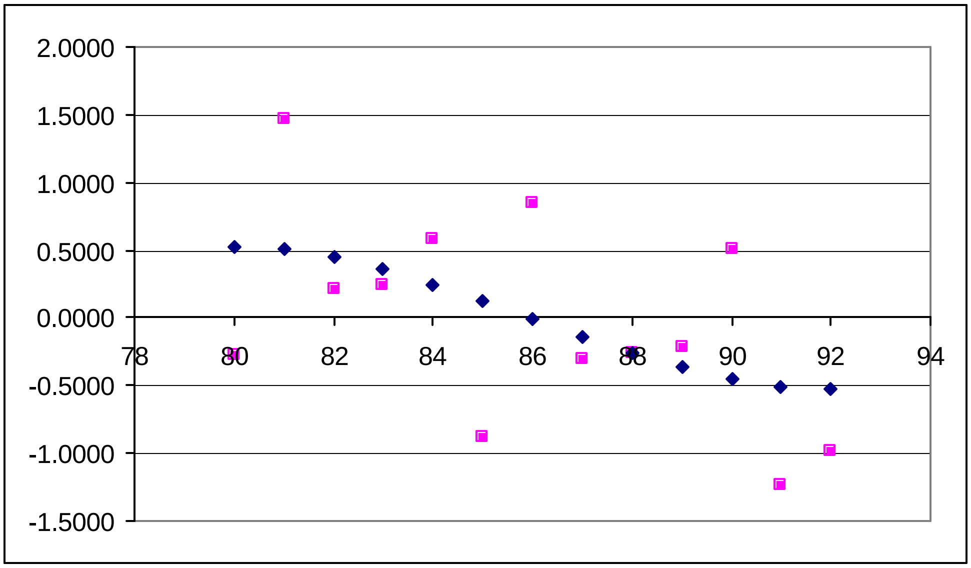


Table 8. Modeled mean annual catch, escapement, number of directed fisheries, and number of population extinctions, in 1,000 runs of 40 years each, at different management thresholds, for Skagit summer/fall chinook. Threshold with maximum catch is bolded.

Skagit Summer/Fall Chinook

Management Threshold	Mean Catch	Mean Escapement	Number of Directed Fisheries	Population Extinctions
10000	12943	9430	29190	7
11000	13003	9706	27435	6
12000	13053	10000	25565	4
13000	13083	10290	24338	4
14000	13094	10579	23167	1
15000	13094	10885	21783	0
16000	13084	11189	20599	0
17000	13066	11484	19480	0
18000	13044	11780	18493	0
19000	13006	12085	17348	0
20000	12961	12386	16243	0

Table 9. Modeled mean annual catch, escapement, number of directed fisheries, and number of population extinctions, in 1,000 runs of 40 years each, at different management thresholds, for Skagit spring chinook. Threshold with maximum catch is bolded.

Skagit Spring Chinook

Management Threshold	Mean Catch	Mean Escapement	Number of Directed Fisheries	Population Extinctions
1500	1569	1664	28056	0
1600	1578	1692	27244	0
1700	1586	1724	26317	0
1800	1592	1755	25323	0
1900	1597	1785	24441	0
2000	1598	1812	23483	0
2100	1598	1838	22558	0
2200	1596	1860	21732	0
2300	1592	1880	20922	0
2400	1587	1898	20145	0
2500	1582	1916	19499	0

Derivation of exploitation rate objectives

Summer / fall chinook

The management objectives for Skagit summer/fall include a recovery exploitation rate that insures, while maintaining fishing opportunity, that harvest will not impede recovery, and low abundance thresholds that guard against abundance falling below the point of instability (Hayman 1999a; 2000a; 2000b). Recovery exploitation rate objectives were developed to meet the following criteria:

1) The percentage of escapements less than the critical abundance (i.e. escapement) threshold increases by less than 5 percentage points relative to the baseline (i.e., in the absence of fishing mortality).

2) Escapements at the end of 25 years exceed the rebuilding escapement threshold at least 80% of the time; **or** the percentage of escapements less than the rebuilding threshold at the end of 25 years differs from the baseline by less than 10 percentage points.

The critical abundance threshold is defined as that which would result in a 5 percent probability that the management unit would become extinct (i.e. fall below 100) at the end of ten years. Since a satisfactory method to calculate critical escapement has not been developed, escapement equal to 5 percent of the stock replacement level was chosen (Hayman 1999a). Replacement escapement is based on the current productivity of the management unit, and therefore incorporates parameters that define the Ricker stock / recruit functions for Skagit units, and recent freshwater and marine survival. For the summer / fall unit, the critical escapement level is 1,165 (Hayman 2000a and 2000b).

The rebuilding escapement threshold is that current level for which there is a 99 percent probability that the run will persist at viable levels. Put another way, if current exploitation rates and freshwater and marine survival conditions were maintained, the probability that the run would go extinct (i.e., fall below 100) at the end of 100 years would fall below one percent. The rebuilding escapement threshold for summer / fall chinook was computed by simulating the population dynamics for 100 years, given a recent average brood year exploitation rate and age composition of escapement, for a range of initial escapement levels. Simulations were replicated 2,000 times, until an initial escapement resulted in extinction in fewer than 1 percent of those replicate runs (Hayman 1999a; 2000b). The rebuilding escapement threshold is 4,700 for the summer/fall unit

With the critical and rebuilding escapement levels established, the population dynamics of the summer / fall Skagit unit was simulated for 25-year periods into the future. The simulation model incorporated the average age composition and age-specific escapement of the units, and randomly or cyclically varying productivity and management error parameters. Each model run used an input exploitation rate, and was replicated 2000 times. The probabilities of exceeding the recovery escapement level, or falling below the critical escapement level, at the end of the simulation period were computed for each run from the 2000 outcomes. A range of exploitation rates, from 0 to 80 percent, were simulated to determine the maximum exploitation rate at which the conservation criteria were met (Hayman 1999a; 2000b). The Washington co-managers have set a rebuilding exploitation rate ceiling of 5 percent for the Skagit summer/fall management unit, as estimated from coded-wire tag recoveries. This management objective was developed from productivity functions characteristic of brood years of Skagit chinook, and was translated into an annual exploitation rate, that is output from the FRAM model, of 50% (Table 4). This exploitation rate objective was set to be 82 percent of the mean rate from fishing years 1989-1993 for summer/fall chinook (Hayman 2000c).

Low abundance thresholds ("crisis escapement levels") were also established for the summer/fall management unit. These thresholds are defined as the pre-season forecast escapement for which there is a 95 percent probability that the actual escapement will be above the point of instability, given management error and uncertainty about what level the point of instability is (Hayman 1999a;2000b). The derivation of these thresholds takes into account the difference between forecast and observed escapement in previous years, and variance of the spawner-recruit parameters used to calculate the point of instability, thereby reducing the probability of actual escapement falling below the actual point of stock instability. The derivation involved varying the preseason forecast until the area of overlap between the management error distribution curve and the uncertainty curve about the point of instability is less than 5% of the error distribution curve (Hayman 2000b).

In low-abundance years, when projected spawning escapement (from the FRAM model) fall to the lower thresholds, fisheries managers will implement further conservation measures in fisheries to reduce mortality, as described in Section 3 and Appendix C. For the summer/fall management unit, the low abundance threshold is 4,800. For the summer/fall unit, low abundance thresholds have been developed for each component population, so that forecast weakness in any one population may trigger the more conservative harvest regime. The low abundance thresholds for Upper Skagit summers, Lower Sauk summers, and Lower Skagit falls are 2,200, 400, and 900, respectively (Hayman 2000a).

The escapement of individual summer/fall populations may be projected from the aggregate escapement, which is output from the simulation model, in proportion to brood year escapement for each population, or in proportion to estimated age-3 and age-4 adults recruited from their brood-year escapement. Survival rates to compute recruitment will be those implied by the Ricker spawner / recruit function for each population.

Spring chinook

Population	***Modeled CET***	***Modeled RET***	A&P RER	FRAM RER
Suiattle	170	400	50%	41%
Upper Sauk	130	330	46%	38%
Cascade	170	Data insufficient to derive a spawner-recruit analysis. RERs for other Skagit spring populations will be used as surrogate		
Spring MU	470[4]	990	47%	38%

Introduction

The rebuilding exploitation rate (RER) is the highest allowable ("ceiling") exploitation rate for the population under normal conditions of stock abundance. This rate is designed to meet the objective that, compared to a hypothetical situation of zero harvest impact, the impact of harvest at this rate will not significantly impede the opportunity for the population to grow towards the recovery goal. Fisheries are then managed to not exceed the ceiling rate. Recovery will require changes to harvest, hatchery, and habitat management. However, our task involves examining only the impacts of harvest on survival and recovery within the context of actions that are occurring in the other sectors affecting listed salmon. Therefore, we evaluate the RER based on Monte Carlo projections of the near-term (25 years) future performance of the population under current productivity conditions, i.e., assuming that the impact of hatchery and habitat management actions remain as they are now. The RER will be periodically evaluated to see if the actions taken in hatchery and habitat management, or changes in natural environmental

[4] In order to account for management error and uncertainty, the spring chinook LAT in this plan will remain at 576 (Hayman 2000b).

conditions would require revisions of our assumptions about productivity or capacity. The RER is defined as the rate that would result in escapements unlikely to fall below a critical escapement threshold (CET) and likely to rebuild above a rebuilding escapement threshold (RET). All sources of fishing-related mortality are included in the assessment of harvest.

There are two phases to the process of determining an RER for a population. The first, or model fitting phase, involves using recent data from the target population itself, or a representative indicator population, to fit a spawner-recruit relationship representing the performance of the population under current conditions. Population performance is modeled as

$$R = f(S, \mathbf{e}),$$

where S is the number of fish spawning in a single return year, R is the number of adult equivalent recruits[5], and **e** is a vector of environmental, density-independent correlates of annual survival.

Several data sources are necessary for this: a time series of natural spawning escapement, a time series of total recruitment, age distributions for both of these, and time series for the environmental correlates of survival. In addition, one must assume a functional form for f, the spawner-recruit relationship. Given the data, one can numerically estimate the parameters of the assumed spawner-recruit relationship to complete the model fitting phase.

The second, or projection phase, of the analysis involves using the fitted model in a Monte Carlo simulation to project the probability distribution of the near-term future performance of the population assuming that current conditions of productivity continue. Besides the fitted values of the parameters of the spawner-recruit relationships, one needs estimates of the probability distributions of the variables driving the population dynamics, including the process error (including first order autocorrelation) of the spawner-recruit relationship itself and each of the environmental correlates. Also, since fishing-related mortality is modeled in the projection phase, one must estimate the distribution of the deviation of actual fishing-related mortality from the intended ceiling. This is termed “management error” and its distribution, as well as the others are estimated from available recent data.

We used the viability and risk assessment procedure (VRAP)(N. Sands, in prep.) for the projection phase. For a series of target exploitation rates the population is repeatedly projected for 25 years. From the simulation results we computed the fraction of years in all runs where the escapement is less than the CET and the fraction of runs for which the average of the spawning escapements in years 21-25 is greater than the RET. Target exploitation rates for which the first fraction is less than 5% and the second fraction is greater than 80% (or less than 10% than would have occurred without harvest) are considered acceptable for use as ceiling exploitation rates for harvest management. These are the RERs.

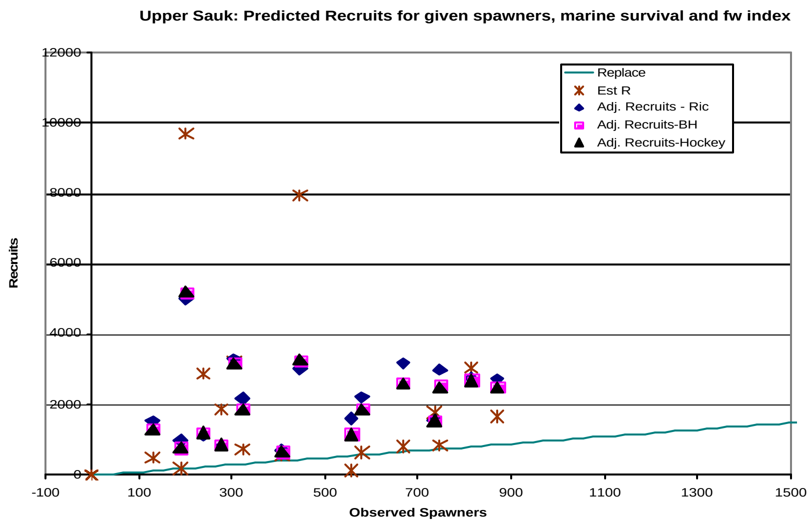
[5] Equivalently, this could be termed “potential spawners” because it represents the number of fish that would return to spawn absent harvest-related mortality.

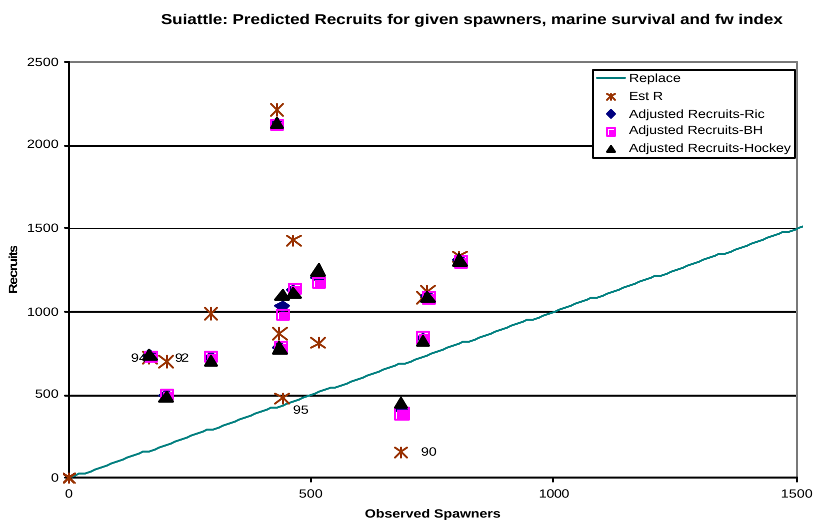
MODEL FITTING PHASE

General

To derive the Suiattle and Upper Sauk spring chinook RERs, we examined the 1981 to 1997 brood years. Uncertainty about data quality of escapement and fishing rates, and residual analyses that indicated a change in system productivity, precluded use of data before 1980. After adjusting for environmental factors, there was no evidence of depensation in the data (Figures 3a and 3b). The 1997 brood year was the last year for which data were available to conduct complete cohort reconstruction.

Figures 3a and 3b. Upper Sauk (1a) and Suiattle (1b) spring chinook recruits adjusted for marine and freshwater environmental conditions





The symbols marked Adj. Recruits (-Bev, -Ric, and –Hoc) in the above figures denote the recruits that would have been produced without the influence of the environmental correlates that drive year to year survival. This allows us to look at the effect of spawners only on the number of recruits produced. We need to remove the effects of other factors, such as the environment, if we want to look for possible depensation which is a function of the number of spawners. Adjusted recruits are calculated for each year as follows:

$$\text{Adjusted recruits} = \frac{\text{Recruits}}{\text{(Annual Environmental Factor/Average Environmental Factor)}}$$

$$\text{Annual Environmental Factor} = (\text{Marine survival index}^\wedge c)(e^{(d*\text{freshwater flow})})$$

$$\text{Average Environmental Factor} = \sum_{year=1}^{t} Annual\_Environmental\_Factor \Big/ t$$

Where c and d are constants from the spawner-recruit relationship

Escapement estimation methods changed in 1994. Although the two methods result in different escapement estimates in any one year, preliminary comparisons of the two methods do not indicate a consistent difference. There was some concern that because the correlation between the old and new method was weaker for the Upper Sauk than for the Suiattle population, it might preclude use of the data to derive an RER for the Upper Sauk spring population. For the Suiattle, the coefficient of variation of the escapement estimates made before this method change is approximately the same as the coefficient of variation of the estimates since 1994, which indicates comparable measurement accuracy in both time periods; in contrast, the greater coefficient of variation in the Upper Sauk before 1994 indicates that measurement error in the Upper Sauk was probably greater before 1994 than since that time (Table 10).

Table 10. Average number of spawners with standard deviation and coefficient of variation (CV) for three time periods.

	Cascade	Upper Sauk	Suiattle
1952-1974			
average		1225	825
st dev		917	378
Cv		75%	46%
autocorrel		0.35	0.27
1975-1993			
average	192	540	546
st dev	84	384	234
Cv	44%	71%	43%
autocorrel		0.22	0.16
1994-2002			
average	284	309	385
st dev	151	138	158
Cv	53%	45%	41%
autocorrel		0.39	(0.37)

While more variable than those of the Suiattle, the Upper Sauk escapements correlated with independent estimates of marine survival, both before and after the change in escapement estimation methods in 1994. This suggests that the estimates prior to 1994 provide useful information about the behavior of the population. If the data were random, one would not expect any correlation with marine survival, and, in fact, when this assumption was tested, the randomized data had no correlation with any marine survival indices (probability of recruitment fit from random data = 96.2-99.9%)(N. Sands, memo to Skagit RER workgroup, 9/2/03). For the Upper Sauk data, since the information is used to derive the productivity parameter for the spawner-recruit models, we also looked to see if the ratio of recruits/spawner (productivity) was significantly different depending on which escapement estimation method was used. Examination of the 1989-1997 data did not indicate a significant difference in the slopes (t-stat = -1.5; prob = 0.1<x< 0.2) or intercepts (t-stat = 1.34; prob = 0.2) of the relationship between spawners and the natural log of recruits/spawner using the old and new escapement estimates. Therefore, we concluded that we did not have sufficient data to demonstrate that the spawner-recruit relationship for the Upper Sauk spring population would be significantly different depending on the escapement estimation methodology used. Therefore, we used the available escapement data (1981-1993 using peak live and dead counts, 1994-1997 using redd counts) to derive the spawner-recruit parameters for the Upper Sauk population (Table 11). When sufficient data is available using the current method based on cumulative redd counts, the RERs will be revised based on that method.

Table 11. Comparison of R/S values under the escapement estimation methods used before and after 1994. The 1989 brood year would be the first returns affected since they would return as 5 year olds in 1994.

	Spawners		Recruits		R/S estimates		Difference
Brood yr	old	new	old	new	old	new	(oldR/S-newR/S)
1989	668	668	1325	821	2.0	1.2	0.8
1990	557	557	659	146	1.2	0.3	0.9
1991	747	747	4282	852	5.7	1.1	4.6
1992	580	580	844	656	1.5	1.1	0.3
1993	323	323	711	749	2.2	2.3	-0.1
1994	574	130	498	496	0.9	3.8	-2.9
1995	1115	190	191	193	0.2	1.0	-0.8
1996	1079	408	553	551	0.5	1.4	-0.8
1997	264	305	3193	3212	12.1	10.5	1.6
1989-97 geomean	596	379	897	589	1.5	1.6	
1989-97 minimum	264	130	191	146	0.2	0.3	
1989-97 maximum	1,115	747	4,282	3,212	12.1	10.5	
1989-97 st. deviation	293	215	1,407	920	3.8	3.2	

Fishery Rates

Fishery rates for both populations were based on the Skagit spring yearling chinook hatchery indicator stock. Although the stock also has a significant fingerling component (41% and 50% on average for the Suiattle and Upper Sauk, respectively), there are only four years (three consecutive) of available exploitation rate data for the fingerling component; too few to define a spawner-recruit relationship. Preliminary analysis indicates there may be differences between yearling and fingerling exploitation rate patterns, but the data is insufficient to determine with any certainty the direction and magnitude of those differences. We considered using fingerling data from the Nooksack early populations, but that population has a much lower percentage of naturally-occurring yearlings and a different harvest pattern, so there was a great deal of uncertainty about whether the Nooksack population would be representative. A Skagit spring chinook fingerling hatchery indicator stock has been established and the co-managers' are collecting data on fingerling exploitation rate patterns. We will re-examine the data for differences in exploitation rate patterns when several more years of data are available. The hatchery indicator stock is used to represent the natural component also because the natural component is not tagged.

The Pacific Salmon Commission Chinook Technical Committee (CTC) CWT exploitation rate analysis for the Skagit spring indicator stock by age was used for brood years 1981 to 1996, ages 2-4 for brood year 1997 and ages 2-3 for brood year 1998. The 1997 age 5+ fishery rate was based on an average of the 1995-96 rates and the 1998 ages 4-5+ were based on an average of the 1996-1997 rates because the current CTC CWT exploitation rate analysis is not complete for these ages for these brood years. For the purposes of the analysis, fishing rates through brood year 1997 were used since this is the most recent brood year for which we have the most available information. Fishery rates will continue to be updated as data become available.

Maturation Rates

Maturation rates were derived from age data collected from scales from the spawning grounds combined with the age-specific fishing rates described above. Age data taken from scales sampled from the spawning grounds were available for return years 1986-90 and 1992-2001 for the Suiattle, and 1986, 1992-95 and 1997-2001 for the Upper Sauk population (WDFW and SSC data 2002). However, we identified two potential concerns that should be taken into account when using the data: 1) age 2 fish are generally underrepresented in spawning ground samples for several reasons: e.g., carcasses decay faster, the smaller body size makes them more susceptible to being washed downstream, they are less visible to samplers; and 2) only eight years for the Suiattle and five years for the Upper Sauk had a sufficient number of samples to use. The age structure for other years was extrapolated from the average brood year age composition of the years that met the sample size criterion to reconstruct brood year and calendar year escapements by age. The age structure is then adjusted to minimize the difference between both the estimated calendar year escapements and the observed calendar year escapements, and the estimated brood year escapements and the observed brood year escapements for each year for which data are not available. Scale samples collected from areas immediately adjacent to the hatchery were excluded because the presence of hatchery fish was assumed to be substantial. Both yearling and fingerling age data were used in order to represent the full range of life histories present in the basin.

Hatchery Effectiveness/Hatchery Contribution to Natural Spawning

The coded-wire tag indicator stock program is the only hatchery production of Skagit spring chinook in the Skagit basin. Straying of hatchery fish onto the spawning grounds from either inside or outside the basin has been negligible based on spawner survey information (WDF et al. 1993, Skagit RER Workgroup 2003). Therefore, hatchery effectiveness is not considered an issue in the derivation of spawner-recruit parameters for the Skagit spring chinook populations.

Spawner-recruit Models

The data were fitted using three different models for the spawner recruit relationship: the Ricker (Ricker 1954, as referenced in Ricker 1975), Beverton-Holt (Beverton and Holt 1957, as referenced in Ricker 1975), and hockey stick (Barrowman and Meyers 2000). The simple forms of these models were augmented by the inclusion of environmental variables correlated with brood year survival. A wide variety of marine and freshwater covariates were evaluated and the ones with the best correlations to estimated recruits/spawner were chosen for further analysis. For marine survival we tried several indices of survival based on chinook coded-wire tag groups from: several Canadian hatcheries in Georgia Strait; several Washington coastal hatcheries; North Puget Sound hatcheries only; South Puget Sound hatcheries only, an aggregate of groups from throughout Puget Sound; Hood Canal hatcheries only; and an aggregate of Puget Sound spring chinook hatcheries. We also evaluated the spawner-recruit function assuming marine survival does not influence the relationship. The other environmental correlate, associated with survival during the period of freshwater residency, was the maximum daily average October 1-February 28 stream flow during the fall and winter of spawning and incubation from the 1) Sauk River USGS gauge near Sauk (gauge # 12189500), 2) the Whitechuck gauge (gauge # 12186000, which is actually on the Sauk just upstream from the Whitechuck), and 3) the Mount Vernon gauge (gauge # 12200500). For the Upper Sauk, we also evaluated the level of spring releases from the Marblemount Hatchery, and the peak instantaneous flow from October to September at the Sauk River gauge (# 12189500). During the time period that escapement and fishing rates data were available, we evaluated the spawner-recruit relationship for three time periods: 1981-1997, 1984-97 and 1986-1997. The spawner-recruit relationship, after adjusting for environmental conditions, appeared relatively constant based on an analysis of the residuals. The results, detailed in Sands (2003), are summarized in Tables 3 and 4, with parameter estimates shown in Tables 5 and 6. A good fit was defined as one with probability of less than 5% for escapement and less than 20% for recruits of being a random fit.

Equations for the three models are as follows:

$$(\mathbf{R} = \mathbf{aS}e^{-\mathbf{bS}})(\mathbf{M}^{\mathbf{c}}e^{\mathbf{dF}}) \qquad \text{[Ricker]}$$

$$(\mathbf{R} = \mathbf{S}/[\mathbf{bS} + \mathbf{a}])(\mathbf{M}^{\mathbf{c}}e^{\mathbf{dF}}) \qquad \text{[Beverton-Holt]}$$

$$(\mathbf{R} = min[\mathbf{aS},\mathbf{b}])(\mathbf{M}^{\mathbf{c}}e^{\mathbf{dF}}) \qquad \text{[hockey stick]}$$

In the above, M is the index of marine survival and F is the freshwater correlate.

Table 12. Results of the spawner-recruit relationship fits for various marine and freshwater covariates for the Suiattle spring chinook population. For each run, the best S/R function fit is noted.

Years	Marine Survival Index	Freshwater Discharge	Model Fit (% esc, % recruit)
1981-97	N. Puget Sound cycle	Sauk max daily ave. Oct-Feb	0, 1
	Puget Sound cycle	Sauk max daily ave. Oct-Feb	0, 0
	Puget Sound cycle	Whitechuck max daily ave	Same as Sauk
	Puget Sound cycle	Mt. Vernon max daily ave	Same as Sauk
	Georgia Strait cycle	Sauk max daily ave. Oct-Feb	0, 2
1984-97	N. Puget Sound cycle	Sauk max daily ave. Oct-Feb	2, 4
	Puget Sound cycle	Sauk max daily ave. Oct-Feb	0, 3
	Puget Sound cycle	Whitechuck max daily ave	Same as Sauk
	Puget Sound cycle	Mt. Vernon max daily ave	Same as Sauk
	Georgia Strait cycle	Sauk max daily ave. Oct-Feb	
1986-97	N. Puget Sound cycle	Sauk max daily ave. Oct-Feb	
	Puget Sound cycle	Sauk max daily ave. Oct-Feb	0, 25
	None	**Sauk max daily ave. Oct-Feb**	**0, 11**

Table 13. Results of the spawner-recruit relationship fits for various marine and freshwater covariates for the Upper Sauk spring chinook population. For each run, the best S/R function fit is noted.

Years	Marine Survival Index	Freshwater Discharge	Model Fit (% esc, % recruit)
1981- 97	**Puget Sound cycle**	**Sauk max daily ave. Oct-Feb**	**0,3**
	Puget Sound cycle	Whitechuck max daily ave	Same as Sauk
	Puget Sound cycle	Marblemount spring releases	0,2
	Puget Sound cycle	Instantaneous Sauk Peak Oct-Sep	0,1
	N. Puget Sound cycle	Instantaneous Sauk Peak Oct-Sep	0,1
	Hood Canal ave.	Instantaneous Sauk Peak Oct-Sep	0,15
	Georgia Strait cycle	Sauk max daily ave. Oct-Feb	0,7
1985-97	Puget Sound cycle	Whitechuck max daily ave	0,9
1986-97	Puget Sound cycle	Whitechuck max daily ave	1,16
	Georgia Strait cycle	Sauk max daily ave. Oct-Feb	3,21
	Hood Canal ave.	Instantaneous Sauk Peak Oct-Sep	2,47

The model fits were evaluated based on the size of the predictive error (MSE), probability of the model being fit by random for escapement data and recruits, the ability of the model to estimate productivity at low abundance and the reasonableness of the model's predicted performance at higher escapement levels, relative to our observations. As seen from Tables 12 and 13, most of the model runs met the criteria for a low probability of resulting from random fit.

For the Suiattle population, the model with the lowest probability of a random fit was the model using the Puget Sound cycle for the marine index and the Sauk maximum daily average winter freshwater flow during 1981-97. However this model and several others did a poor job of estimating productivity at low abundance even though the probability of random fit was low. The model for the 1986-97 period assuming no influence from marine survival and using the Sauk maximum daily average winter freshwater flow had the best overall combination of a low predictive error, probability of random fit and estimate of productivity at low abundances compared with the other model runs (Figures 2 and 3, Tables 5a and 5b). In particular, the data points were well distributed along the spawner-recruit curve, both the predicted and observed data fit the curve defined by the spawner-recruit relationship well, and there was little difference among the three spawner-recruit functions (Figure 3). Finally, while both the 1981-97 and 1986-97 relationships estimated capacity at about 800 spawners, the 1981-97 relationship implied considerable redd superimposition between 400 and 800 spawners which has not been observed in the field with escapements in this range.

For the Upper Sauk population, there were two models with the lowest probability of a random fit: the peak Oct-Feb winter freshwater flow combined with 1) the North Puget Sound fall fingerling cycle marine index; and 2) the Puget Sound cycle marine index, during 1981-97. However, the data points for the models for the period 1981-97 using the Puget Sound marine index were better distributed along the spawner-recruit curve (Figures 4 and 5). There was little difference in the fit among the models using the Puget Sound cycle marine index or their estimates of the escapement at maximum sustained yield[6] (Tables 6a and 6b). The model using the Puget Sound cycle for the marine index and the Sauk maximum daily average winter flow for

[6] The Beverton-Holt function did a poor job of describing productivity at low escapement regardless of the model.

the 1981-97 period was used as the representative model of this group for purposes of deriving the RER since it fit well and it matched the freshwater variable used for the Suiattle .

Figure 4. Comparison of observed and predicted recruitment for the Suiattle spring population, brood years 1981-97 data, the Puget Sound cycle marine index and Sauk maximum daily average winter flows, under three different models of the spawner-recruit relationship. The corresponding spawner-recruit parameters are listed in Table 5a.

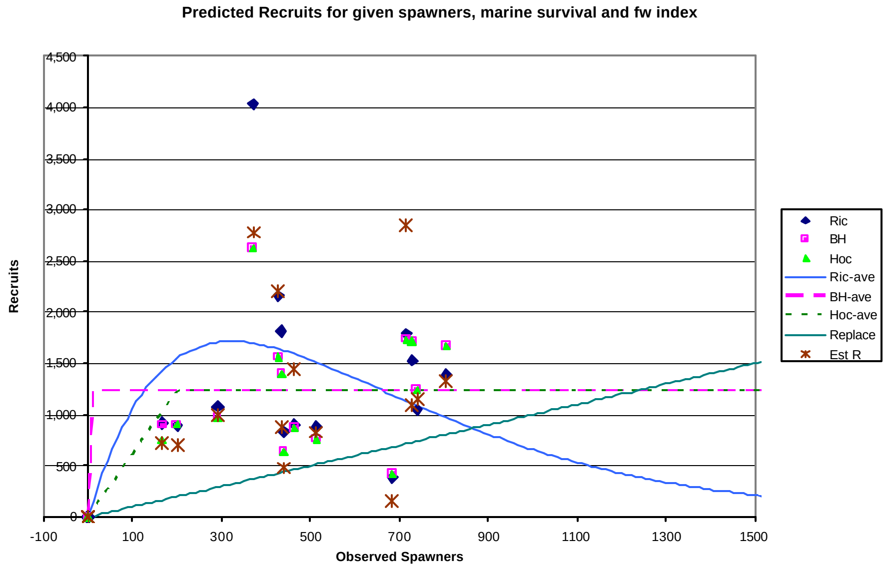


Figure 5. Comparison of observed and predicted recruitment for the Suiattle spring population, brood years1986-97 data, no marine index and Sauk maximum daily average winter flows, under three different models of the spawner-recruit relationship. The corresponding spawner-recruit parameters are listed in Table 5b

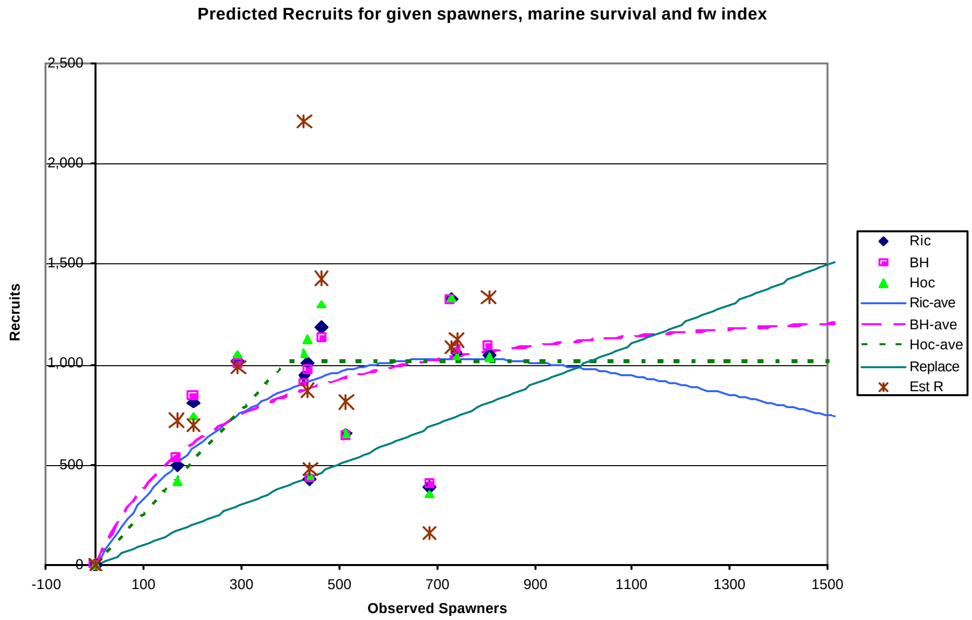


Table 14a (left) and 14b (right). Results of spawner-recruit analysis for the Suiattle using different time periods and environmental covariates.

	14a			14b		
Marine Index	**Puget Sound cycle**			**none**		
Freshwater variable	**Sauk maximum daily ave. Oct-Feb**			**Sauk maximum daily ave. Oct-F**		
calendar years esc. compared	**1986-1997**			**1991-1997**		
brood years used	**1981-1997**			**1986-1997**		
Parameter Estimates With Smallest S	**Ric**	**Bev**	**Hoc**	**Ric**	**Bev**	**Hoc**
a - productivity	27.8956	0.0000	13.1729	6.5805	0.1112	4.6642
b - Spawners	0.003293	0.000380	2,648	0.001351	0.000417	1,835
c - Marine	0.8132	0.7634	0.7604	0.9800	0.9800	0.9800
d - Freshwater	-0.000012	-0.000017	-0.000017	-0.000022	-0.000021	-0.000024
SSE	0.287	0.707	0.705	0.019	0.024	0.016
MSE (esc)	0.036	0.088	0.088	0.005	0.006	0.004
autocorrelation in error	0.090	0.018	0.027	-0.034	-0.147	0.040
R - esc	0.949	0.866	0.867	0.992	0.989	0.993
F(3,8)	24.122	8.035	8.063	118.032	93.600	138.566
PROBABLITIY	0.0%	0.8%	0.8%	0.0%	0.0%	0.0%
MSE (recruits)	0.272	0.274	0.270	0.215	0.227	0.195
autocorrelation in error	0.028	-0.068	-0.059	-0.163	-0.127	-0.220
R - recruits	0.822	0.750	0.748	0.636	0.614	0.684
F(3,13)	9.014	5.579	5.506	3.060	2.728	3.959
PROBABLITIY	0.6%	2.3%	2.4%	15.6%	17.9%	11.3%
Ave.Pred. Error	1020	1218	1219	469	480	440

	Ric	**Bev**	**Hoc**	**Ric**	**Bev**	**Hoc**
slope at origin, intrinsic prod.	27.90	1000.00	13.17	6.58	9.00	4.66
average MS*FW factor	0.75	0.66	0.65	0.57	0.59	0.55
cv MS/FW	61/17	57/23	57/24	0/34	0/32	0/36
adjusted productivity at origin	20.79	657.36	8.61	3.78	5.31	2.58
replacement level	920	1,730	1,730	980	1,160	1,020
capacity = spawners for max recruits	300	1,730	200	740	1,420	400
max recruits	2,320	1,730	1,730	1,030	1,420	1,020
MSY spawners	260	10	210	410	350	400
MSY recruits	2,300	1,730	1,730	890	810	1,020
MSY ER	0.89	0.99	0.88	0.54	0.57	0.61
ave ER last 3yrs	0.72	0.72	0.72	0.69	0.69	0.69

Figure 6. Comparison of observed and predicted recruitment for the Upper Sauk spring population, brood years 1981-97 data, the North Puget Sound cycle marine index and peak instantaneous Oct-Sep flow at the Sauk gauge, under three different models of the spawner-recruit relationship.

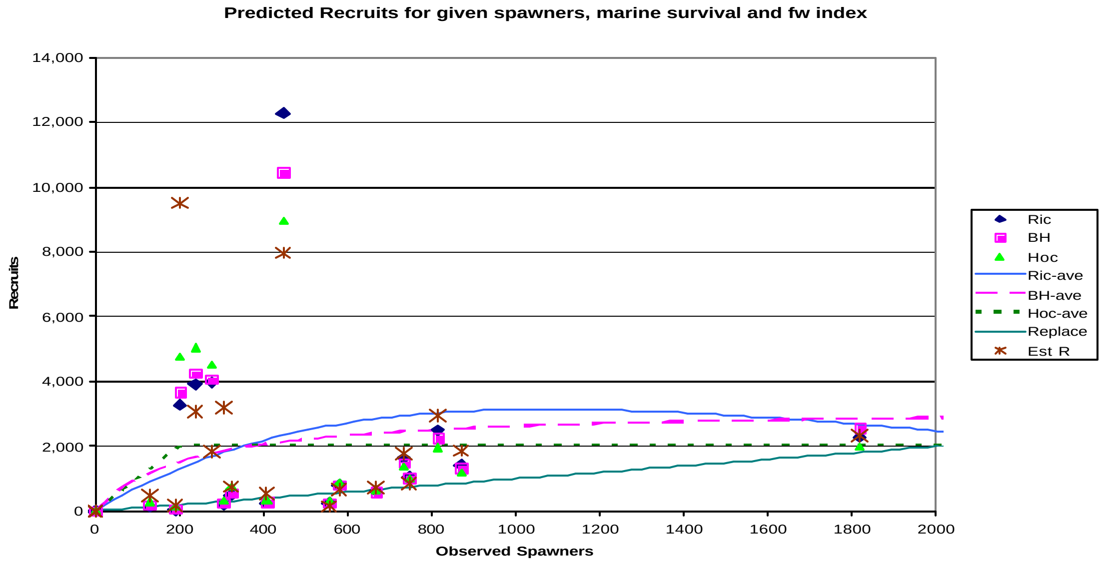


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Figure 7. Comparison of observed and predicted recruitment for the Upper Sauk spring population, brood years1981-97 data, the Puget Sound cycle marine index and peak instantaneous Oct-Sep flow at the Sauk gauge, under three different models of the spawner-recruit relationship. The corresponding spawner-recruit parameters are listed in Table 6a.

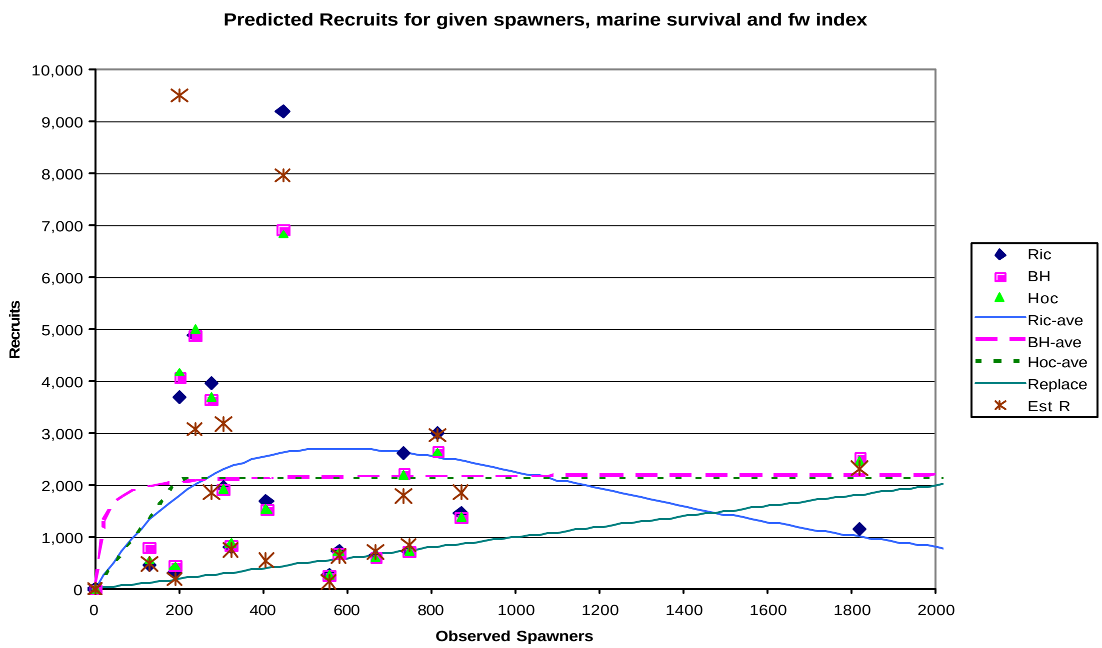


Table 15a (left) and 15b (right). Results of spawner-recruit analysis for the Upper Sauk using different freshwater environmental covariates.

	15a			15b		
marine index	**Puget Sound cycle**			**Puget Sound cycle**		
freshwater index	**inst. peak Oct-Sep. winter flow**			**Sauk maximum daily average winter flow (Oct-Feb)**		
= calendar years esc. compared	**1986-1997**			**1986-1997**		
= brood years used	**1981-1997**			**1981-1997**		
	Ric	**Bev**	**Hoc**	**Ric**	**Bev**	**Hoc**
a - productivity	24.5562	0.0035	20.7467	21.3694	0.0037	17.1128
b - Spawners	0.001721	0.000232	4,191	0.001745	0.000282	3,457
c - Marine	1.2134	1.0926	1.0766	1.1330	1.0135	0.9991
d - Freshwater	-0.000021	-0.000020	-0.000020	-0.000026	-0.000022	-0.000022
SSE	0.216	0.253	0.238	0.119	0.259	0.245
MSE (esc)	0.027	0.032	0.030	0.015	0.032	0.031
autocorrelation in error	0.736	-0.362	-0.276	0.481	-0.184	-0.166
R - esc	0.974	0.969	0.971	0.986	0.969	0.970
F(3,8)	48.666	41.413	44.111	90.778	40.732	42.923
PROBABLITIY	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
MSE (recruits)	0.350	0.325	0.308	0.418	0.401	0.388
autocorrelation in error	0.147	0.429	0.375	0.163	0.410	0.372
R - recruits	0.763	0.808	0.812	0.693	0.721	0.723
F(3,13)	6.040	8.131	8.385	4.002	4.700	4.749
PROBABLITIY	1.9%	0.8%	0.7%	5.2%	3.6%	3.5%
Ave.Pred. Error	1919	1769	1752	2145	2094	2087

	15a Ric	15a Bev	15a Hoc	15b Ric	15b Bev	15b Hoc
slope at origin, intrinsic prod.	24.56	286.46	20.75	21.37	268.20	17.11
average MS*FW factor	0.52	0.51	0.51	0.59	0.61	0.61
cv MS/FW	87/36	79/35	78/35	82/33	74/30	73/30
adjusted productivity at origin	12.68	147.43	10.60	12.57	163.52	10.39
replacement level	1,480	2,200	2,140	1,450	2,160	2,100
capacity = spawners for max recruits	580	2,220	200	570	2,160	200
max recruits	2,710	2,220	2,140	2,650	2,160	2,100
MSY spawners	480	180	220	460	150	220
MSY recruits	2,670	2,040	2,140	2,590	1,990	2,100
MSY ER	0.82	0.91	0.90	0.82	0.92	0.90
ave ER last 3yrs	0.72	0.72	0.72	0.72	0.72	0.72
set survival	0.16	0.18	0.19	0.19	0.23	0.23
adj MSY sp	330	90	200	330	90	200
adj MSY recruits	730	670	760	760	710	790
adj MSY ER	0.55	0.87	0.74	0.57	0.87	0.75

Critical Abundance Threshold

The critical abundance threshold (CAT) represents a boundary below which uncertainties about population dynamics increase substantially. If sufficient stock-specific information is available, we can use the population dynamics relationship to define this point. Otherwise, we use alternative population-specific data, or general literature-based guidance. In this case, the CAT is 170 and 130 for the Suiattle and Upper Sauk spring chinook populations, respectively, and 470 for the spring MU, using the smallest previously observed escapement from which there was a greater than 1:1 return per spawner. Other escapements in this range have also generated returns per spawner of greater than one (Skagit RER Workgroup 2003). NOAA Fisheries has also provided some guidance on the range of critical thresholds in its document, *Viable Salmonid Populations* (McElhaney et al. 2000). The VSP guidance suggests that effective population sizes of less than 500 to 5,000 per generation, or 125 to 1,250 per annual escapement, are at increased risk. The CATs of 130 and 170 fall within the lower end of this range, reasonable for a small population (Upper Sauk: 1980-2002 range = 130-1,818, average = 459; Suiattle: 1980-2002 range = 167-1094, average =503).

It is important to distinguish between the CAT used in this RER calculation, and the LAT used in this harvest management plan. Although the Suiattle and Upper Sauk modeled CET numbers are the same as their LATs (see Tables 1 and 3 of the harvest management plan), they don't represent the same thing. The modeled CAT is an assumed point of instability; however, because the CAT's used in the RER calculation are escapement levels from which the observed return per spawner was greater than 1:1, it is likely that these modeled CAT levels are in fact well above the true points of instability, a bias that will build conservatism into the calculated RER. The LAT, on the other hand, is a trigger point below which additional management actions are taken to prevent escapement from falling below the true CAT. The LATs that were used for the Skagit summer/fall populations and the spring management unit during the last 3 years were calculated as the preseason escapement forecasts for which there is a 5% probability that the post-season escapement number will be less than the point of instability (Hayman 2000a; Hayman 2000b). Interestingly, using the spawner-recruit parameters derived from this RER analysis, the LAT for Suiattle chinook was calculated as 170 (assuming a quasi-extinction threshold of 63), which is the same as the modeled CAT number that was derived using the 1:1 return rate as the criterion. The calculated LAT for Upper Sauk chinook would be 250, which is higher than the number calculated from the 1:1 return rate criterion; however, because of the greater variance about the Upper Sauk spawner-recruit relation, the estimated probability that an escapement of 130 would be below the point of instability was unrealistically high, given that we have observations that indicate that it in fact is not below this point. Thus, for Upper Sauk chinook, we set the LAT at the same value as the modeled CAT (130). Assuming that the Upper Sauk point of instability is 72 (as calculated from the spawner-recruit parameters), and the past observed range of management error, the probability that a forecasted escapement of 130 would result in an observed escapement below the point of instability was only 0.2%. For the Skagit spring MU, the calculated LAT was 576 (Hayman 2000b), which is over 100 chinook higher than the CET assumed in this analysis (470). Because there is nothing in the LAT calculation that appears to contradict our observations (e.g., there is a very low probability that an escapement of 470, the lowest observed escapement with a return rate greater than 1:1, is below the point of instability), we retained 576 as the LAT in this harvest management plan.

Rebuilding Escapement Threshold

The RET represents a higher abundance level that would generally indicate recovery or a point beyond which ESA type protections are no longer required. Again, because we are isolating the effects of harvest, the RET in this context represents an escapement level consistent with estimates of the current productivity and capacity of the Upper Sauk and Suiattle spring chinook populations. The RET is the smallest escapement level such that the addition of one additional spawner would be expected to produce less than one additional future recruit under current conditions of productivity[7]. This level is also known as the maximum sustainable yield (MSY) escapement. The rebuilding threshold varies with the assumed freshwater covariate and also with the particular form of the spawner-recruit relationship.

For the Suiattle, using the maximum daily flow in the Sauk River from October through February, we derived the RET for each spawner-recruit function. These values were: 410 – Ricker, 350 – Beverton-Holt, and 400 – hockey stick (Table 5a). Since all three models performed similarly (Table 2), we propose to use the average of these estimates as the RET. This average is 400 natural origin spawners (rounding to the nearest 100 spawners).

For the Upper Sauk, using the maximum daily flow in the Sauk River from October through February and the Puget Sound cycle marine index, we derived the RET for each spawner-recruit function. These values were: 460 – Ricker and 220 – hockey stick, under the 1981-97 marine survival rates. However, in our VRAP runs (see next section) we assumed that marine survival in the near future would be more similar to the generally lower rates estimated for 1988-95, for which the RET values were: 330 – Ricker and 200 – hockey stick (Table 6b). For reasons explained in the next section, we discarded the hockey stick analysis and used the Ricker value, 330, as the RET for Upper Sauk. The Beverton-Holt spawner-recruit function did a poor job of estimating productivity at low abundance and, therefore, was not used to estimate a RET.

It is extremely important to recognize that the RET is not an escapement goal but rather a level that is expected to be exceeded most of the time ($\geq$ 80%) under the RER. It is also the case that, should the productivity conditions for the population improve, the RET and the corresponding RER will increase under improved conditions. However, since we will not be able to detect these changes immediately, the RER under current conditions provides a conservative approach because it assumes conditions are poorer than may actually exist. Should conditions improve, the probability of exceeding the RET using the RER computed for current conditions will also increase over the probability computed under current conditions. Thus the RET serves as a step in the progression to recovery which will occur as the contributions from all sectors are realized.

Rebuilding Exploitation Rate Derivation

We projected the performance of the Suiattle and Upper Sauk spring population at target exploitation rates in the range of 0 to 0.80 at intervals of 0.02 using the fitted values of a, b, c, and d (see model equations above) for the Upper Sauk spawner-recruit models, and using the fitted values of a, b, and d for the 3 Suiattle models (which had no marine survival parameter; hence, no c value). As described above, for the Suiattle, we used the 1986-97 brood year model run using the Sauk monthly maximum average flow during the winter, and no marine survival parameter. For the Upper Sauk, we used the 1981-97 brood year model run using the Puget Sound marine cycle index and the Sauk maximum daily average flow during the winter. The freshwater environmental correlate (maximum daily average flow) was projected using the average and

[7] An alternative definition of RET, i.e., the initial escapement level from which there is less than 1% probability that the unit will go extinct in 100 years, was used to set the RER for the Skagit summer/fall and spring management units during the last 3 years (Hayman 1999; Hayman 2000a; Puget Sound Indian Tribes and WDFW 2001; Puget Sound Indian Tribes and WDFW 2003). However, the programming necessary to use this definition for the Skagit spring populations has not been completed, so RETs that use this definition for the Skagit spring populations were not calculated.

variance observed for the 1981-1997 period. For the Upper Sauk, the marine survival environmental correlate (Puget Sound cycle) was projected using the average and variance observed for the 1988-95 period, a period of low marine survival. West coast salmon have been experiencing a period of low marine survival. Although there are preliminary indications that marine conditions are improving, it has not yet been confirmed for Puget Sound. The CETs were 170 and 130 for the Suiattle and Upper Sauk, respectively, derived as described above. The RETs were the MSY escapement levels (also described above) adjusted for environmental conditions. When adjusted for projected environmental conditions the RETs for the Upper Sauk population were: 330 – Ricker and 200 – hockey stick. Since marine survival did not influence the spawner-recruit relationship, no adjustment for environmental conditions to the RET was required for the Suiattle population.

For each combination of spawner-recruit relationship and exploitation rate we ran 1000 25-year projections. Estimated probabilities of exceeding the RET were based on the number of simulations for which the average of the spawning escapements in years 21-25 exceeded the RET. Estimated probabilities of falling below the CET were based on the number of years (out of the total of 25,000 individual years projected for each target exploitation rate for a particular spawner-recruit relationship) that the spawning escapement fell below the CET. For each spawner-recruit relationship the sequence of Monte Carlo projection running through the target exploitation rate range from 0 to 0.80 started with the same random number seed so that the results for the different spawner-recruit models would be comparable.

Detailed results of these projections are in Tables 18 to 21, and summarized results are in Tables 16 and 17. For the Suiattle, the indicated target exploitation rates are 0.48 – Ricker, 0.52 – Beverton-Holt, and 0.51 – hockey stick. Since all three models performed similarly, we propose to use the average of these values as the target rebuilding exploitation rate. This average is 0.50, rounding down to the nearest whole percentage exploitation rate.
For the Upper Sauk, the target exploitation rates that meet the RER criteria are 0.46 – Ricker and 0.62 – hockey stick. A comparison of the habitat in the areas used by the three Skagit spring populations indicated the productivities of the three Skagit spring populations should be similar based on habitat characteristics and land use (B. Hayman, memo to Skagit RER workgroup, 7/15/03). In addition, a VRAP analysis of the Skagit spring management unit (all three spring populations combined) indicated an RER of 0.47 (Tables 18 - 21; N. Sands memo to Skagit RER workgroup, Summary of Skagit springs results, 7/15/03). Since the Ricker target exploitation rate of 0.46 was more similar to the RER for the Suiattle (0.50) and to the Skagit management unit, it was chosen as the RER for the Upper Sauk spring chinook population.

To make the RER compatible with the fishery model used in fishery planning (the FRAM model), the RERs derived from data in the A&P tables were converted to a FRAM equivalent RER using a simple regression between the exploitation rate estimates from the A&P table and post season exploitation rate estimates derived from FRAM. Using this conversion, the FRAM RERs used for annual preseason fishery planning purposes were 0.41 and 0.38 for the Suiattle and Upper Sauk, respectively.

Table 16. Results of the VRAP projections of the Suiattle chinook stock under current conditions showing the indicated target exploitation rate for each form of the spawner-recruit relationship.

	Target	#fish	%runs	%yrs	%runs	1st	LastYrs
Model	ER	Mort.	extinct	<critical	end>rebuilding	Year	Ave.
Ricker	0.48	577	0	0.3	82.3	474	578
Beverton-Holt	0.52	601	0	0.7	80.9	451	500
Hockey-Stick	0.51	635	0	0.4	81.0	460	552

Table 17. Results of the VRAP projections of the Upper Sauk chinook stock under current conditions showing the indicated target exploitation rate for each form of the spawner-recruit relationship.

	Target	#fish	%runs	%yrs	%runs	1st	LastYrs
Model	ER	Mort.	extinct	<critical	end>rebuilding	Year	Ave.
Ricker	0.46	516	0.2	0.5	80.5	620	505
Hockey-Stick	0.62	646	0.9	3.7	85.0	432	327

Table 18**.** Summary of projections of the Suiattle spring chinook population at different target exploitation rates for three different forms of the spawner-recruit relationship.

	Pr (final esc > rebuilding threshold) %				Pr (annual esc < critical threshold) %		
Target ER	**B-H**	**Ricker**	**Hockey-St**		**B-H**	**Ricker**	**Hockey-St**
0.00	100	99.7	100		0	0.1	0
0.02	100	99.8	100		0	0.1	0
0.04	100	99.9	100		0	0	0
0.06	100	99.5	100		0	0	0
0.08	100	99.8	100		0	0.1	0
0.10	100	99.8	100		0	0	0
0.12	100	99.9	100		0	0	0
0.14	100	99.8	100		0	0	0
0.16	100	99.8	100		0	0	0
0.18	100	99.7	100		0	0	0
0.20	100	99.8	100		0	0	0
0.22	100	99.5	99.9		0	0.1	0
0.24	100	99.7	100		0	0	0
0.26	100	99.5	99.9		0	0	0
0.28	100	99.6	99.9		0	0	0
0.30	100	99	99.9		0	0.1	0
0.32	100	98.7	99.3		0	0	0
0.34	99.7	98.9	99		0	0	0
0.36	99.7	97.4	99		0	0	0
0.38	99.7	96.5	98.2		0	0	0
0.40	99.6	95.8	96.5		0	0.1	0
0.42	97.9	92.4	97.1		0.1	0.1	0
0.44	96	87.6	96.1		0.1	0.1	0
0.46	94.5	87.5	93.7		0.1	0.1	0.1
0.48	91.8	**82.3**	90.1		0.2	0.3	0.1
0.50	87.8	74.7	**84.3**		0.4	0.4	0.3
0.52	**80.9**	66.7	**78.7**		0.7	0.8	0.5
0.54	73.3	56	71		1.3	1.3	0.8
0.56	65.7	46.8	57.5		1.9	1.7	2
0.60	53.5	35.4	47.6		3.2	3.2	2.9
0.62	38	23.3	34		**5.6**	**5.6**	**5.4**
0.64	27.3	14.1	22.1		9.1	9.6	9.8
0.66	16.6	5.8	10.9		13.6	15.3	16.8
0.68	9.4	4.1	3.7		21	23.7	28.4

Table 19. Summary of projections of the Upper Sauk spring chinook population at different target exploitation rates for three different forms of the spawner-recruit relationship.

	Pr(final esc > rebuilding threshold)%		Pr(ann. Esc. < critical threshold) %	
Target ER	**Ricker**	**Hockey-St**	**Ricker**	**Hockey-St**
0.00	98.5	100.0	0.3	0.0
0.02	99.2	100.0	0.3	0.0
0.04	97.8	100.0	0.3	0.0
0.06	97.5	100.0	0.2	0.0
0.08	99.3	100.0	0.2	0.0
0.10	98.3	100.0	0.2	0.0
0.12	98.7	100.0	0.2	0.0
0.14	98.1	100.0	0.3	0.0
0.16	98.8	100.0	0.1	0.0
0.18	97.5	100.0	0.2	0.0
0.20	97.5	100.0	0.2	0.0
0.22	96.9	100.0	0.2	0.0
0.24	96.9	100.0	0.1	0.0
0.26	96.2	100.0	0.1	0.0
0.28	96.1	100.0	0.2	0.0
0.30	96.0	100.0	0.1	0.0
0.32	94.7	100.0	0.2	0.0
0.34	95.0	100.0	0.2	0.0
0.36	93.3	100.0	0.2	0.0
0.38	92.2	100.0	0.3	0.0
0.40	92.4	99.7	0.2	0.0
0.42	88.9	99.9	0.3	0.0
0.44	86.1	99.8	0.3	0.0
0.46	**80.5**	99.7	0.5	0.0
0.48	76.7	99.4	0.7	0.0
0.50	74.2	99.0	0.7	0.0
0.52	69.4	97.6	1.1	0.0
0.54	62.9	96.5	1.6	0.1
0.56	55.5	95.9	2.3	0
0.58	48.9	95.4	**3.4**	0
0.60	35.9	89.8	5.6	0.4
0.62	27.8	**85.0**	8.1	0.9
0.64	21.4	78.5	11.4	**2.6**
0.66	12.0	65.4	16.9	6.5

Table 20. Results of spawner-recruit analysis for the Skagit spring management unit using different freshwater environmental covariates.

calendar years esc. compared **1989-1997**
brood years used **1984-1997**
Parameter Estimates With Smallest SSE

	Ric	Bev	Hoc
a - productivity	9.6393	0.0255	5.7893
b - Spawners	0.000759	0.000220	4,185
c - Marine	0.6669	0.5731	0.5839
d - Freshwater	-0.000009	-0.000009	-0.000008
SSE	0.126	0.108	0.107
MSE (esc)	0.025	0.022	0.021
autocorrelation in error	-0.189	-0.060	0.036
R - esc	0.942	0.951	0.951
F(3,5)	13.108	15.642	15.776
PROBABLITIY	1%	1%	1%
MSE (recruits)	0.463	0.426	0.429
autocorrelation in error	0.372	0.428	0.332
R - recruits	0.746	0.764	0.765
F(3,10)	4.175	4.663	4.708
PROBABLITIY	8%	7%	6%
Ave.Pred. Error	2054	2026	1996

	Ric	Bev	Hoc
slope at origin, intrinsic prod.	9.64	39.25	5.79
average MS*FW factor	0.87	0.85	0.87
cv MS/FW	48/15	42/15	43/14
adjusted productivity at origin	8.41	33.54	5.01
replacement level	2,810	3,780	3,620
capacity = spawners for max recruits	1,320	3,880	720
max recruits	4,080	3,880	3,620
MSY spawners	990	540	720
MSY recruits	3,930	3,200	3,610
MSY ER	0.75	0.83	0.80
ave ER last 3yrs	0.73	0.73	0.73

Table 21. Summary of projections of the Skagit spring chinook management unit at different target exploitation rates for the Ricker spawner-recruit relationship.

Target ER	Pr(final esc > rebuilding threshold)%	Pr(ann. Esc. < critical threshold) %
0.00	98.20	0.7
0.02	98.00	0.5
0.04	98.2	0.6
0.06	97.90	0.5
0.08	98.80	0.5
0.10	97.70	0.5
0.12	97.70	0.4
0.14	98.00	0.4
0.16	97.60	0.5
0.18	98.00	0.4
0.20	97.40	0.4
0.22	96.90	0.4
0.24	97.90	0.3
0.26	97.40	0.3
0.28	95.60	0.4
0.30	96.10	0.4
0.32	95.60	0.4
0.34	95.00	0.3
0.36	92.10	0.3
0.38	92.70	0.4
0.40	91.60	0.4
0.42	88.50	0.4
0.44	88.20	0.6
0.46	**83.60**	0.6
0.48	**78.30**	0.7
0.50	76.20	1.0
0.52	71.60	1.3
0.54	66.20	1.8
0.56	58.10	1.7
0.60	51.90	2.5
0.62	39.90	3.3
0.64	36.30	**5.3**
0.66	25.10	7.9
0.68	15.70	12.2

The ceiling exploitation rates defined in this plan, which are intended to maximize long-term harvestable numbers and prevent extinction for the Skagit spring and summer/fall management units separately, are consistent with a “no jeopardy” ruling. The jeopardy standards themselves were explicitly used to calculate those rates, and the calculated ceiling rates are comparable to the rates on Skagit summer/fall chinook that were evaluated and approved in the Northern Fisheries Biological Opinion (NMFS 2000), which, depending on abundance, ranged from about 50 to 70 percent. Additional conservatism, beyond that evaluated in the Northern BO, is also provided. Critical abundance threshold escapement levels, below which additional actions would be required, are established for both the spring and summer/fall chinook management units separately, and for each of the three summer/fall populations proposed in WDFW & WWTIT (1994). The intent of this Plan is to take actions that prevent extinction of individual populations, while maximizing long-term harvestable numbers and achieving ESA jeopardy standards for the two Skagit wild chinook management units

During pre-season fishery planning, the impacts from a proposed fisheries management regime will be simulated, and escapement projected, based on the forecast abundance of all contributing chinook units (including those from British Columbia, the Washington coast, and the Columbia River, as well as those from Puget Sound). If the projected escapement of either management unit, or of any Skagit summer/fall or spring population falls below their low abundance threshold, further management actions will be triggered to reduce fishing mortality, as described in Chapter 5 and Appendix C. The FRAM fisheries simulation model, which is currently in use, estimates escapement for the Skagit summer/fall management unit, but that management unit total may be resolved into component stocks in proportion to their forecasted total abundance.

An analysis of how this regime would have functioned if it had been applied in previous years indicates that the exploitation rates would generally have been significantly lower than observed, and that the management response to critical status would have been triggered in two of the recent years (R. Hayman, Skagit System Cooperative pers comm.)

Data gaps

Priorities for filling data gaps to improve understanding of stock / recruit functions or population dynamics simulations necessary to testing and refining harvest management objectives include:

- Consistent release of coded-wire tagged fingerling summer and fall chinook to enable direct assessment of harvest distribution, and estimation of harvest exploitation rates and marine survival rates;.
- Estimates of natural-origin smolt abundance from spring chinook production areas.
- Estimates of estuarine and early-marine survival for fingerling and yearling smolts.
- Limiting factors on yearling chinook abundance

Stillaguamish River Management Unit Status Profile

Component Stocks

Stillaguamish summer chinook
Stillaguamish fall chinook

Geographic description

The Stillaguamish River management unit includes summer and fall stocks which are distinguished by differences in their spawning distribution, migration and spawning timing, and genetic characteristics. The summer stock, a composite of natural and hatchery-origin supplemental production, spawns in the North Fork, as far upstream as RM 34.4 but primarily between RM 14.3 and 30.0, and in the lower Boulder River and Squire Creek. Spawning also occurs in French, Deer, and Grant creeks, particularly when flows are high. The fall stock, which is not enhanced or supplemented by hatchery production, spawns throughout the South Fork and the mainstem of the Stillaguamish River (WDF et al. 1993), and in Jim Creek, Pilchuck Creek, and lower Canyon Creek. Despite the small overlap in spawning distribution, it is likely that the two stocks are genetically distinct.

Allozmye analysis of the summer stock show it to be most closely related to spring and summer chinook stocks from North Puget Sound, and the the Skagit River summer stocks in particular. The fall stocks align most closely with South Sound MAL, which includes Green River falls and Snohomish River summer and falls.

Life History Traits

Summer run adult enter the river from May through August. Spawning begins in late August, peaks in mid-September, and continues past mid-October. Fall chinook enter the river much later – in August and September. The peak of spawning of the fall stock occurs in early to mid-October, about three weeks later than the peak for the summer stock. The age composition of mature Stillaguamish River summer chinook, based on scales collected from 1985 – 1991 was as follows: 4.9% age-2, 31.9% age-3, 54.7% age-4, and 8.5% age-6 (WDF 1993 cited in HGMP). Juvenile summer chinook produced in the Stillaguamish River primarily (95%) emigrate as sub-yearlings (WDF 1993 cited in HGMP).

Status

WDF et al. (1993) classified both the summer and fall stocks as depressed, due to chronically low escapement. Degraded spawning and rearing habitat currently limit the productivity of chinook in the Stillaguamish River system (PFMC 1997). After analyzing the trends in spawning escapement through 1996, the PSC Chinook Technical Committee concluded that the stock was not rebuilding toward its escapement objective (CTC 1999).

Aggregate spawning escapement for Stillaguamish summer/fall chinook has averaged 1,341 (geometric mean) over the period 1997 – 2001. From 1988 through 1995 escapement ranged from 700 to 950 (except 1991), and since 1995 has ranged from 1100 to over 1600. The geometric mean of escapement in the last five years (1998 - -2002) was 1429, which was higher than the mean of 1009 from the preceding five years (Myers et al. 1998). From 1985 – 1991 the average escapements of summer and fall chinook were 879 and 145, respectively (WDF et al.

1993). In the last five years (1998-2002) escapement to the South Fork ranged from 226 – 335), while escapement to the North Fork ranged from 845 to 1403 . Escapement to the North Fork has comprised an average of 81% of total escapement since 1997 (K. Rawson, Tulalip DNR, pers comm., February 10, 2003).

Table 1. Spawning escapement of Stillaguamish summer/fall chinook, 1993-2002.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
North Fork	583	667	599	993	930	1292	845	1403	1066	1253
South Fork	345	287	223	251	226	248	253	243	283	335
Total	928	954	822	1244	1156	1540	1098	1646	1349	1588

The total annual abundance of Stillaguamish summer/fall chinook for the period 1979 – 1995, estimated as potential escapement (i.e. the number of chinook that would have escaped to spawn absent fishing mortality), ranged from 1,300 to 2,500 without showing a clear positive or negative trend (PSSSRG 1997). However, the productivity, as indexed by the trend in MSY exploitation rate, declined substantially through this period.

The summer chinook supplementation program, which collects broodstock from the North Fork return, was initiated in 1986 as a Pacific salmon Treaty indicator stock program, and its current objective is to release 200,000 tagged fingerling smolts per year. Most releases are into the North Fork, via acclimation sites; relatively small numbers of smolts have been released into the South Fork. This supplementation program is considered essential to the recovery of the stock, so these fish are included in the listed ESU. The program contributes substantially to spawning escapement in the North Fork.

Harvest distribution

Recoveries of coded-wire tagged North Fork Stillaguamish summer chinook provide an accurate description of recent harvest distribution. Northern fisheries in Alaska and British Columbia account for 73 percent of total harvest mortality (Table 2). Washington ocean fisheries account for 4 percent. Washington sport fisheries account for 24 percent of total fisheries mortality.

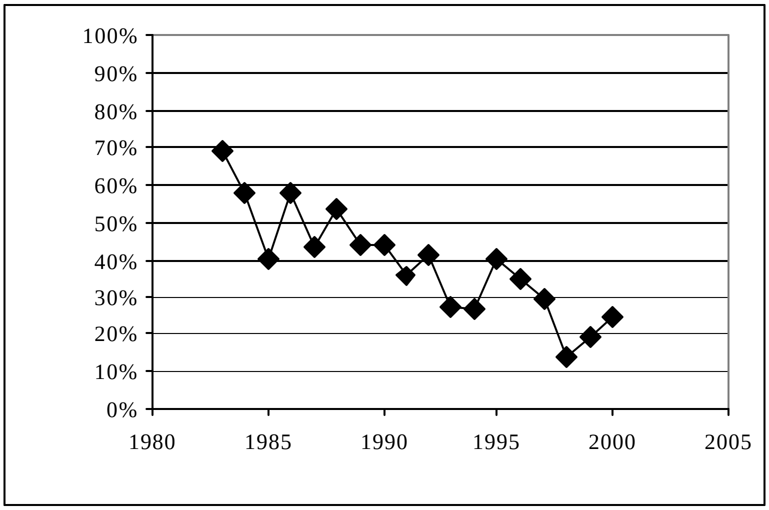
Table 2. The harvest distribution of Stillaguamish River summer chinook, expressed as an average proportion of annual adult equivalent harvest mortality for 1996 - 2000 (CTC03-1 in press)). Update with 2001??

Alaska	B.C.	WashingtonT roll	Puget Sound Net	Washington sport
26.7%	46.3%	0.5%	2.8%	23.8%

Exploitation rate trends:

Post-season FRAM runs, incorporating actual catch in all fisheries and actual abundance, indicate that total fishery-related, adult equivalent, exploitation rates for Stillaguamish chinook have fallen 64 percent, from 1983 – 1987 to 1998 – 2000.

Figure 1. Total adult equivalent fishery exploitation rate of Stillaguamish chinook from 1983 – 2000, estimated by post-season FRAM runs.



Management Objectives

The management guidelines for Stillaguamish chinook include an exploitation rate objective and a critical escapement threshold. The exploitation rate objective is the maximum fraction of the production from any brood year that is allowed to be removed by all sources of fishery-related mortality, including direct take, incidental take, and non-landed mortality. The exploitation rate is expressed as an adult equivalent rate, in which the mortality of immature chinook is discounted relative to their potential survival to maturity.

Analysis specific to Stillaguamish summer chinook was completed to develop the exploitation rate objective to reflect, to the extent possible, the current productivity of the stock. Brood year recruitment (i.e., number of recruits per spawner) was estimated, for brood years 1986 through 1993, by reconstructing the total abundance of natural origin chinook that were harvested or otherwise killed by fisheries, or escaped to spawn. The resulting brood year recruitment rates were partitioned into freshwater and marine survival rates. The future abundance (i.e. catch and escapement) of the stock was simulated for 25 years, using a simple population dynamics model, under total fishery exploitation rates that ranged from 5 percent to 60 percent. In the model, production from each year's escapement was subjected to randomly selected levels of freshwater and marine survival, and randomly selected levels of management error. Each model run (i.e. for each level of exploitation rate) was replicated one thousand times, and the set of projected population abundances analyzed to determine the probability of achieving the management objectives. The simulation for Stillaguamish summer chinook, across a range of exploitation rates (Table 3), indicated that total exploitation rates below 0.35 met the recovery criteria.

Table 3. Summary of results of 1,000 runs of the simulation model at each exploitation rate.

Exploitation Rate	Probability of Falling below critical	Probability of recovery	Median Escapement ratio	Median Escapement
0.00	1%	96%	2.75	3,597
0.05	1%	96%	2.81	3,377
0.10	1%	96%	2.76	3,165
0.15	2%	95%	2.66	2,964
0.20	2%	95%	2.56	2,758
0.25	3%	93%	2.57	2,418
0.30	4%	92%	2.48	2,210
0.35	6%	92%	2.46	1,920
0.40	7%	91%	2.29	1,686
0.45	11%	87%	2.14	1,444
0.50	17%	80%	1.92	1,180
0.60	41%	52%	1.04	648
0.70	73%	12%	0.27	259
0.80	94%	0%	0.02	55

The fishery management objectives for the 2000 management year was to realize an exploitation rate that, if imposed consistently over a future time interval

- would not increase the probability that the stock abundance would fall below the critical escapement threshold, after 25 years, by more than five percentage points higher than were no fishing mortality to occur; and

- would result in at least an 80 percent of greater probability of the stock recovering (i.e. escapement exceeding the current level) after 25 years.

Stock recovery, for this analysis, was defined as the average spawning escapement for the final three years in the simulation period exceeding the average for the first three years in the simulation period (Rawson 2000).

At the present time, there is very little information concerning the productivity of the Stillaguamish fall stock other than the fact that the average abundance of this stock has been approximately 50% of the Stillaguamish summer stock based on relative escapement. Incorporating this lower estimate of abundance, and assuming the same productivity (i.e. recruitment rates), the simulation model predicted that exploitation rates below 35% met the first management objective. The probability of rebuilding at this exploitation rate was 96%. This analysis indicates that a target exploitation rate of 0.35 would also be appropriate for the Stillaguamish fall stock.

The Washington co-managers have set an exploitation rate guideline of 0.25, as estimated by the FRAM simulation model, for the Stillaguamish chinook management unit. According to the simulation model this level of exploitation results in a 4 percent risk of the stocks falling below the critical escapement threshold of 500, and affords a 92 percent probability of recovery (i.e., that spawning escapement will exceed the current average level).

The low abundance threshold for North Fork Stillaguamish chinook is 500 natural-origin spawners. Reconstruction of the total brood abundance of adult Stillaguamish chinook suggests that escapements of 500 (+/- 50) can result in recruitment rates ranging from two to five adults per spawner (Rawson 2000). The genetic integrity of the stock may be at risk and depensatory mortality factors may affect the stock when annual escapement falls below this threshold to 200 (NMFS BO 2000). The critical threshold for South Fork Stillaguamish chinook is undetermined pending further analysis of data. The low abundance threshold for the Stillaguamish management unit is based on the 1996-2002 average fraction of the natural escapement for the years 1996-2002 that was in the North Fork. This average was .813 (range: .770 - .852). Thus a management unit escapement of 500/.813 = 615 would, on average, include 500 North Fork fish. The range of management unit escapement thresholds computed this way is 586 to 649. Based on this, we have selected a low abundance threshold of 650 for the Stillaguamishmanagement unit. Whenever spawning escapement is projected to be below this level, fisheries will be managed to either achieve the critical exploitation rate ceiling , or exceed the low abundance threshold .

Data gaps

Priorities for filling data gaps to improve understanding of stock / recruit functions or population dynamics simulations necessary to testing and refining harvest management objectives include:

- Spawning escapement estimates that include variance for summer and fall stocks
- Estimates of natural-origin smolt production (freshwater survival to the estuary)

Snohomish River Management Unit Status Profile

Component Stocks

The stock structure of summer/fall chinook in the Snohomish basin is based on the report of the Puget Sound TRT (2001) suggesting that there are two populations of summer/fall chinook in the Snohomish basin. The comanagers have reviewed this report along with additional information, and have tentatively concluded that the former four-stock structure of Snohomish chinook should be revised to conform to the TRT's population structure.

Summer/fall chinook management unit

Skykomish
Snoqualmie

Geographic description

Skykomish chinook spawn in the mainstem of the Skykomish River, and its tributaries including the Wallace and Sultan Rivers, in Bridal Veil Creek, the South Fork of the Skykomish between RM 49.6 and RM 51.1 and above Sunset Falls (fish have been transported around the falls since 1958), and the North Fork up to Bear Creek Falls (RM 13.1). Relative to spawning distribution in the 1950's, a much larger proportion of summer chinook currently spawn higher in the drainage, between Sultan and the forks of the Skykomish (Snohomish Basin Salmonid Recovery Technical Committee (SBSRTC) 1999). There is some indication that spawning in the North Fork has declined over the last twenty years (Snohomish Basin Salmonid Recovery Technical Committee (SBSRTC) 1999). Fish spawning in Snohomish mainstem and the Pilchuck River are currently considered to be part of the Skykomish stock pending further collection of genetic stock identification data.

Snoqualmie chinook spawn in the Snoqualmie River and its tributaries, including the Tolt River, Raging River, and Tokul Creek.

There is some uncertainty whether a spring chinook stock once existed in the Snohomish system. Suitable habitat may still exist in the upper North Fork, above Bear Creek Falls.

Life History Traits

Summer chinook enter freshwater from May through July, and spawn, primarily, in September, while fall chinook spawn from late September through October. However, fall chinook spawning in the Snoqualmie River continues through November. The peak of spawning in Bridal Veil creek is in the second week of October (i.e. slightly later than the peak for fish spawning in the mainstem of the Skykomish. Natural spawning in the Wallace River occurs throughout September and October (Washington (State). Dept. of Fisheries. et al. 1993).

The age composition of returning Snoqualmie River fall chinook showed a relatively strong age-5 component (28 percent), relative to other Puget Sound fall stocks. Age-3 and age-4 fish comprised 20 and 46 percent, respectively, of returns in 1993 – 1994 (Myers et al. 1998).

Most Snohomish summer and fall chinook smolts emigrate as subyearlings, but, based on scale data, an annually variable, but relatively large, proportion of smolts are yearlings. Of the summer chinook smolts sampled in 1993 and 1994, 33 percent were yearlings (Myers et al. 1998). Based on scale data, 25 to 30 percent of returning fall chinook also showed a stream-type life history (Snohomish Basin Salmonid Recovery Technical Committee (SBSRTC) 1999). No other summer or fall chinook stocks in Puget Sound produces this high a proportion of yearling smolts. Rearing habitat to support yearling smolt life history is vitally important to the recovery of these stocks.

Management Unit / Stock Status

Total natural spawning escapement of Snohomish summer/fall stocks has ranged between 2,700 and 8,200 since 1990, and has exceeded the 1968-1979 average of 5,237 only four times since 1980: in 1998, 2000, 2001, and 2002 (Table 1). However, due in part to reduced exploitation rate, escapement has rebounded from the levels observed in the early 1990s.

Table 1. Natural spawning escapement of Snohomish summer/fall chinook salmon, 1990-2002. Total estimates of natural spawning escapement were provided by WDFW using the escapement estimation method described by Smith and Castle (Smith and Castle 1994). Estimates of the natural origin fraction of the natural escapement are based on recoveries of thermally marked otoliths (Rawson et al. 2001)

Year	Snoqualmie	Skykomish	Total	Nat. Origin
1990	1277	2932	4209	
1991	628	2192	2820	
1992	706	2002	2708	
1993	2366	1653	4019	
1994	728	2898	3626	
1995	385	2791	3176	
1996	1032	3819	4851	
1997	1937	2355	4292	3525
1998	1892	4412	6304	2856
1999	1344	3455	4799	2436
2000	1427	4665	6092	3024
2001	3589	4575	8164	6336
2002	2895	4325	7220	
average	1443	3146	4791	
average %	31.4%	68.6%		

A portion of the natural spawning fish are the survivors of releases from the Wallace River and Bernie Kai-Kai Gobin (Tulalip) facilities. Since 1997 it has been possible to estimate the natural origin portion of the natural escapement because all chinook production at the Bernie Kai-Kai Gobin and Wallace River hatcheries has been thermally mass-marked and there has been comprehensive sampling of natural spawning areas for otoliths (Rawson et al. 2001). In most years the natural origin component of the natural escapement is significantly smaller than the total natural escapement estimate, although in 2001 the natural origin portion alone of the natural escapement was higher than the total natural escapement in any prior year since at least 1980 (Table 1 and state/tribal chinook escapement database).

Harvest distribution and exploitation rate trends:

Assessment of exploitation rate trends for Snohomish summer/fall chinook is difficult because there has been no coded-wire tagged indicator stock representing the management unit. Post-season runs of the FRAM model show a clearly declining trend in annual fishing year exploitation rate over the past two decades (Table 2). These validation runs use the same projection model used in preseason planning, but use post-season estimates of spawning escapement and fishery harvest and non-catch mortality instead of preseason abundance and fishing level predictions. Thus, these runs adjust for observed abundances and fishing levels, but they assume the stock composition of fisheries is the same as the base period stock composition used in the FRAM model.

Table 2. Adult equivalent (AEQ) exploitation rates (ER) by fishing year for the Snohomish summer/fall chinook management unit from post-season runs of the FRAM model for 1983-2000 (April 2003 revision of FRAM validation runs, personal communication, Andy Rankis, NWIFC, and Larrie LaVoy, WDFW) and from pre-season FRAM model predictions for 1999-2003[8]. The ceiling exploitation rate column is the maximum allowable annual AEQ exploitation rate from the management plan that was in effect for the year[9].

	AEQ ER		
Fishing Year	**Postseason**	**Preseason**	**Ceiling ER**
1983	73%		
1984	64%		
1985	55%		
1986	60%		
1987	48%		
1988	66%		
1989	52%		
1990	49%		
1991	52%		
1992	61%		
1993	62%		
1994	50%		
1995	65%		
1996	44%		
1997	29%		
1998	25%		
1999	31%	31%	38%
2000	26%	20%	35%
2001		21%	32%
2002		18%	32%
2003		19%	24%

[8] FRAM runs 99NP, 00NP, 01NP, 02NP, and 03NP.
[9] These are documented in the annual Stillaguamish/Snohomish regional status reports available from Tulalip Fisheries, 7615 Totem Beach Rd., Marysville, WA 98271. Management objectives that were in effect for years before 1999 are also documented in regional status reports for those years.

Table 3. Brood year exploitation rates reported in the Puget Sound Technical Recovery Team's Abundance and Productivity tables for the Skykomish and Snoqualmie chinook populations.

Brood Year	Skykomish	Snoqualmie
1980	86%	86%
1981	88%	87%
1982	84%	77%
1983	68%	67%
1984	82%	83%
1985	75%	74%
1986	76%	74%
1987	70%	69%
1988	76%	78%
1989	74%	75%
1990	67%	59%
1991	54%	39%
1992	56%	61%
1993	61%	64%
1994	54%	54%
1995	46%	38%
1996	51%	44%
1997	46%	43%
1998	48%	46%

Management Objectives

Management objectives for Snohomish summer/fall chinook include an upper limit on total exploitation rate, to insure that harvest does not impede the recovery of the component stocks, and a low abundance threshold (LAT) for spawning escapement to trigger reduced fishing effort under low returns to maintain the viability of the stocks. Fisheries will be managed to achieve a total adult equivalent exploitation rate, associated with all salmon fisheries, not to exceed 24 percent. These impacts include all mortalities related to fisheries, including direct take, incidental take, release mortality, and drop-off mortality.

Lacking direct information on the extent to which the current fisheries regime may disproportionately harvest any single stock, the spawning escapement of each stock will be carefully monitored for indications of differential harvest impact. Average escapement during the period of 1965 – 1976 will be the benchmark for this monitoring (Snohomish Basin Salmonid Recovery Technical Committee (SBSRTC) 1999).

The Puget Sound Salmon Management Plan mandates that fisheries will be managed to achieve maximum sustainable harvest (MSH) for all primary[10] natural management units. The recovery exploitation rate is likely to be lower than the rate associated with MSH under current conditions of productivity, as in the case where recovery involves increasing the current level of productivity. The conservatism implied by the recovery exploitation rate imbues caution against the potential size and age selectivity of fisheries, and the effects of that selectivity on reproductive potential, and potential uncertainty and error in management.

[10] A primary management unit is one for which fisheries are directly management to achieve a particular escapement goal or exploitation rate.

LOW ABUNDANCE THRESHOLD FOR MANAGEMENT

A low abundance threshold of 2,800 spawners (natural origin, naturally spawning fish) for the Snohomish management unit is established (see estimation procedure below) as a reference for pre-season harvest planning. If escapement is projected to fall below this threshold under a proposed fishing regime, extraordinary measures will be adopted to minimize harvest mortality. Directed harvest of Snohomish natural origin chinook stocks, (net and sport fisheries in the Snohomish terminal area or in the river) has already been eliminated. Further constraint, thus, depends on measures that reduce incidental take.

The low abundance threshold for the management unit was derived from critical escapement thresholds for each of the Snoqualmie, and Skykomish populations in a two-step process. Critical escapement thresholds are levels that we don't want to go below under any circumstances. For each population, the critical escapement threshold was determined and then expanded to an adjusted level for management use according to the following formula:

$$E_{man,p} = E_{crit,p} / [(R/S)_{low,p} * (1-RER_{mu})] \qquad [1]$$

Where $E_{man,p}$ is the lower management threshold for population ***p;***
$E_{crit,p}$ is the critical threshold for population ***p;***
$R/S_{low,p}$ is the average of recruits/spawner for population ***p*** under low survival conditions; and
RER_{mu} is the RER established for the management unit

The following describes the $\boldsymbol{E_{man,p}}$ for the Snoqualmie and Skykomish stocks within the Snohomish management unit. The following analysis is based on estimates of natural spawning escapement to the Snohomish system, by population, for the most recent twelve years (Table 1) .

Maximum Exploitation Rate Guideline

INTRODUCTION

The rebuilding exploitation rate (RER) is the highest allowable ("ceiling") exploitation rate for a population under recovery given current habitat conditions , which define the current productivity and capacity of the population. This rate is designed to meet the objective that, compared to a hypothetical situation of zero harvest impact, the impact of harvest under this Plan will not significantly impede the opportunity for the population to grow towards the recovery goal. Since recovery will require changes to harvest, hatchery, and habitat management and since this Plan only addresses harvest management, we cannot directly evaluate the likelihood of this plan's achieving its objective. Therefore, we evaluate the RER based on Monte Carlo projections of the near-term future performance of the population under current productivity conditions, in other words, assuming that hatchery and habitat management remain as they are now and that survival from environmental effects remain as they are now.

We choose the RER such that the population is unlikely to fall below a critical threshhold[11] (CT) and likely to grow to or above a rebuilding escapement threshold (RET). The CT is chosen as the smallest previously-observed escapement from which there was a greater than 1:1 return per

[11] Note that, there are other provisions of this plan that call for further reduction of the exploitation rate ceiling should the abundance be observed or expected to be near the lower threshold. This will provide additional protection against falling below the lower threshold that is not considered in this section, which address only the conditions under which the RER would apply.

spawner, while the RET is chosen as the smallest escapement level such that the addition of one additional spawner would be expected to produce less than one additional future recruit under current conditions of productivity. This level is also known as the maximum sustainable harvest (MSH) escapement. It is extremely important to recognize, though, that under this Plan the RET is not an escapement goal but rather a level that is expected to be exceeded most of the time. It is also the case that, when the productivity conditions for the population improve due to recovery actions, the RET will usually increase (MSH escapement does not increase in the Hockey stick model if productivity and capacity increase together as in eq. 5) and the probability of exceeding the RET using the RER computed for current conditions will also increase over the probability computed under current conditions. Thus the RET serves as a proxy for the true goal of the plan, which can only be evaluated once we have information on likely future conditions of habitat that will result from recovery actions, and hatchery as well as harvest management.

It also follows from the above, given that the likely chance of achieving the RET is greater than 50%, that the actual harvest from the population under this Plan will be less than the maximum sustainable harvest, the amount less being dependent on the likelihood (%) of achieving the RET. All sources of fishing-related mortality are included in the assessment of harvest, and nearly 100% of the fishing-related mortality will be due to non-retention or incidental mortality; only a very small fraction is due to directed fishing on Snohomish populations.

There are two phases to the process of determining an RER for a population. The first, or model fitting phase, involves using recent data from the target population itself, or a representative indicator population, to fit a spawner-recruit relationship representing the performance of the population under current conditions. Population performance is modeled as

$$R = f(S, \mathbf{e}),$$

where S is the number of fish spawning in a single return year, R is the number of adult equivalent recruits[12], and **e** is a vector of environmental, density-independent correlates of annual survival. The purpose of this phase is to be able to predict the recruits from spawners and environmental covariates into the future. What is important here is to simulate a pattern of returns into the future, not predict returns for specific years.

Several data sources are necessary for this analysis: a time series of natural spawning escapement, a time series of total recruitment (obtained from run reconstruction based on harvest and escapement data), age distributions for both of these, and time series for the environmental correlates of survival. In addition, one must assume a functional form for f, the spawner-recruit relationship; in our case three different forms were examined. Given the data, one can numerically estimate the parameters of the assumed spawner-recruit relationship to complete the model fitting phase.

The second, or projection phase, of the analysis involves using the fitted model in a Monte Carlo simulation to predict the probability distribution of the near-term future performance of the population assuming that current conditions of productivity continue. Besides the fitted values of the parameters of the spawner-recruit relationships, one needs estimates of the probability distributions of the variables driving the population dynamics, including the process error (including first order autocorrelation) of the spawner-recruit relationship itself and each of the environmental correlates. Also, since fishing-related mortality is modeled in the projection

[12] Equivalently, this could be termed "potential spawners" because it represents the number of fish that would return to spawn absent harvest-related mortality.

phase, one must estimate the distribution of the deviation of actual fishing-related mortality from the intended ceiling. This is termed "management error" and its distribution, as well as the others are estimated from available recent data.

We used the viability and risk assessment procedure (VRAP, N J Sands, in prep.) for the projection phase. For each trial RER value, the population is repeatedly projected for 25 years. From the simulation results we computed the fraction of years in all runs where the escapement is less than the LAT and the fraction of runs for which the final year's escapement (average of last 3 years) is greater than the UAT. Trial RERs for which the first fraction is less than 5% and the second fraction is greater than 80% are considered acceptable for use as ceiling exploitation rates for management under this plan.

MODEL FITTING PHASE

General

The model used to estimate the spawner recruit parameters uses fishing rate and maturation rate estimates along with the spawning estimates to determine the time series of total recruitment needed.

Preterminal Fishery Rates

Fishery rates were based on an aggregate of Puget Sound summer/fall chinook hatchery indicator stock populations (Stillaguamish, Green, Grovers, George Adams, Nisqually, Samish). Although a new indicator stock tagging program has been implemented to represent Skykomish wild chinook, there is currently no coded-wire-tag (CWT) recovery data available that is directly representative of the Snohomish populations and no direct measure of fishery exploitation on the wild populations. We evaluated two options for estimating fishery rates on the Snohomish populations: 1) an aggregate of Puget Sound summer/fall chinook hatchery coded-wire-tag (CWT) indicator stocks using the Pacific Salmon Commission Chinook Technical Committee (CTC) exploitation rate indicator stock analysis (CTC 1999 for method, Dell Simmons pers. Comm. for most recent data); and 2) estimates from the CTC chinook model (CTC 1999).

Option 1 relies on CWT recoveries from individual years to reconstruct the fishery rates for that year, but is dependent on a consistently high rate of catch and escapement sampling to make precise estimates. After further evaluation, we determined that catch and escapement sampling for most of the populations within the aggregate meet or exceed their target sampling rates in most years. Snohomish populations may not have the same distribution as the populations within the aggregate. Puget Sound summer/fall chinook populations show some similarity in the general trend over time of exploitation in preterminal fisheries. Although it is logical to assume that Snohomish summer/fall populations follow a similar trend with respect to the change over time in the rate of preterminal exploitation, concern remains that the aggregate Puget Sound indicator stocks may not accurately reflect the true exploitation rates of Snohomish populations. Also, the indicator stocks that comprise the aggregate are not likely to represent harvest patterns of yearling outmigrant or "stream type" (Healy 1991). Scale pattern analysis of Snohomish Chinook shows that a significant portion of the return is stream type from both fingerling and yearling populations.

Under Option 2, the CTC model uses CWT recoveries from the Stillaguamish indicator stock during the 1979-1982 base period to estimate fishery exploitation on the Snohomish population in subsequent years so estimates are less subject to year-year variability in sampling rates. The CTC model appears to best reflect the pattern of reduced overall exploitation they expected to see in the early 1990s in response to more restrictive fishing regimes. Again, it is possible that the distribution and exploitation of the Stillaguamish and Snohomish populations are different.

We chose Option 1 because we determined that, for the purposes of deriving an RER, year specific fishery rates would be better than estimates derived from a base period based on a limited number of Stillaguamish CWT recoveries. Option 1, by using an aggregate set of populations, maximizes the use of the available data and smoothes differences in any one year associated with a particular population. Also, we were able to address most of the concerns we had with Option 1. In addition, Therefore, the aggregate was used as a surrogate to represent the Snohomish populations in preterminal fisheries. Fishery rates were derived from the CTC CWT exploitation rate analysis for each population in the aggregate and averaged across all populations for each year for which data were available.

The average CTC CWT exploitation rate analysis for fall indicator stocks by age was used for brood year 1979 to 1994, ages 2-4 for brood year 1995 and ages 2-3 for brood year 1996. The 1995 age 5+ fishery rate was based on an average of the 1993-94 rates. The 1996 ages 4-5+ were based on an average of the 1994-1995 rates because the current CTC CWT exploitation rate analysis is not complete for these ages for these brood years. However, available data for ages 2 and 3 indicate fishery rates were similar in 1994-1996. Fishery rates will continue to be updated as data become available.

Terminal Fishery Rates

Terminal area fisheries include mature chinook harvested in net fisheries throughout Puget Sound and in recreational fisheries in the Snohomish River system and Area 8D. The in-river recreational fishery harvest is partitioned into natural and hatchery-produced components based on the relative magnitudes of the escapement to natural areas and to the Wallace River Hatchery.

The stock composition of the Area 8D recreational and net harvest is estimated using results of recoveries of thermally-marked otoliths from Tulalip hatchery. The otolith recoveries are used to estimate the Tulalip hatchery contribution to this fishery for the brood years from 1997 on (Rawson et al. 2001), which is subtracted from the total catch. The remaining catch is partitioned into components based upon the relative run strengths of the Stillaguamish and Snohomish chinook returns to their rivers. In particular, the Snohomish natural fraction is estimated as the Snohomish natural escapement plus the Snohmish natural portion of the in-river recreational harvest divided by the sum of the escapements to the Stillaguamish and Snohomish Rivers and the in-river harvests of chinook in those rivers. For years before 1997 the procedure is the same, except that the proportional contribution of Tulalip hatchery fish to Area 8D is assumed to be the average of the values measured for 1997-2001.

The stock composition of the Area 8A net harvest is estimated using the relative proportions of all the Stillaguamish/Snohomish stocks passing through Area 8A. Only chinook harvested during the so-called “adult accounting period” of July1 through September 30 are included in this analysis. Other chinook harvested in Area 8A are part of the preterminal fishing rate. In particular, the Snohomish natural fraction is the sum of the Snohomish natural escapement, the Snohomish natural fraction of the in-river harvest, and the Snohomish natural fraction of the 8D harvest, divided by the sum of the total escapement and harvest in both rivers plus the Area 8D harvest and escapement to Tulalip hatchery.

To the three harvest components computed above (in-river, 8D, and 8A) the harvest of mature Snohomish natural chinook in Puget Sound net fisheries outside of Area 8A must be added. This computation was completed using coded-wire tag recoveries by Jim Scott and Dell Simmons of the CTC. The terminal, or mature fishery, fishing rate is then the sum of the harvest in the four components divided by the numerator plus the Snohomish natural escapement.

Maturation Rates

We also considered two options for the maturation rates (the fraction of each cohort that leaves the ocean to return to spawn during the year): 1) maturation rates derived from age data collected from scales and otoliths from the spawning grounds combined with the age-specific fishing rates described above; 2) estimates derived from the CTC model for the Snohomish model population. In general, fish matured at older ages under option 1 than option 2, and no fish matured as two year olds. We decided to use option 1 because it is a more direct measure of the age structure of the spawners and relies on age specific data for the populations.

However, we identified two potential concerns that should be taken into account when using the data: 1) age 2 fish are generally underrepresented in spawning ground samples for several reasons: e.g., carcasses decay faster, the smaller body size makes them more susceptible to being washed downstream, they are less visible to samplers; and 2) only one year, 1989, had a sufficient number of samples to use. The age structure for other years was extrapolated from 1989 by using the 1989 age composition to reconstruct brood year and calendar year escapements by age. The age structure is then adjusted to minimize the difference between the estimated calendar year escapements and the observed calendar year escapements for each year for which data are not available.

Hatchery Effectiveness

No adjustments were made for the relative fecundity of naturally-spawning hatchery-produced fish as compared with natural-origin fish, since there is no available data for the effectiveness of hatchery spawners in the wild when compared with their natural origin counterparts for Puget Sound chinook. For the RER analysis, we assumed all spawners were equally fecund regardless of their origin. This is a conservative assumption since it would tend to underestimate productivity (assuming hatchery fish are less effective) and, therefore, the resulting RER, minimizing the possibility of adopting a harvest objective that was too high (Table 4.)

Table 4. Intrinsic Productivity (MSY Exploitation Rate) by Production Function for the Skykomish chinook population.

Hatchery Effectiveness	**Ricker**	**Beverton-Holt**	**Hockey Stick**
Not Effective	7.58 (49%)	14.14 (65%)	8.07 (77%)
Half as Effective	6.26 (52%)	8.34 (65%)	4.55 (63%)
Equal Effectiveness	5.49 (47%)	6.51 (53%)	3.66 (51%)

Spawner-recruit Models

The data were fitted using three different models for the spawner recruit relationship: the Ricker (Ricker 1975), Beverton-Holt (Ricker 1975), and hockey stick (Barrowman and Myers 2000). The simple forms of these models were augmented by the inclusion of environmental variables correlated with brood year survival. For marine survival we used an index based on the common signal from a several chinook coded-wire tag groups released from Puget Sound hatcheries (J Scott, Washington Department of Fish and Wildlife, personal communication). We tried two indices: one (PS6) used tag groups from throughout Puget Sound; the other (NPS2) used coded wire tags from North Puget Sound hatcheries only. The other environmental correlate, associated with survival during the period of freshwater residency, was the September-March peak daily mean stream flow during the fall and winter of spawning and incubation.

Equations for the three models are as follows:

$(\mathbf{R} = \mathbf{aS}e^{-\mathbf{bS}})(\mathbf{M}^{\mathbf{c}}e^{\mathbf{dF}})$ [Ricker]

$(\mathbf{R} = \mathbf{S}/[\mathbf{bS} + \mathbf{a}])(\mathbf{M}^{\mathbf{c}}e^{\mathbf{dF}})$ [Beverton-Holt]

$(\mathbf{R} = min[\mathbf{aS}, \mathbf{b}])(\mathbf{M}^{\mathbf{c}}e^{\mathbf{dF}})$ [hockey stick]

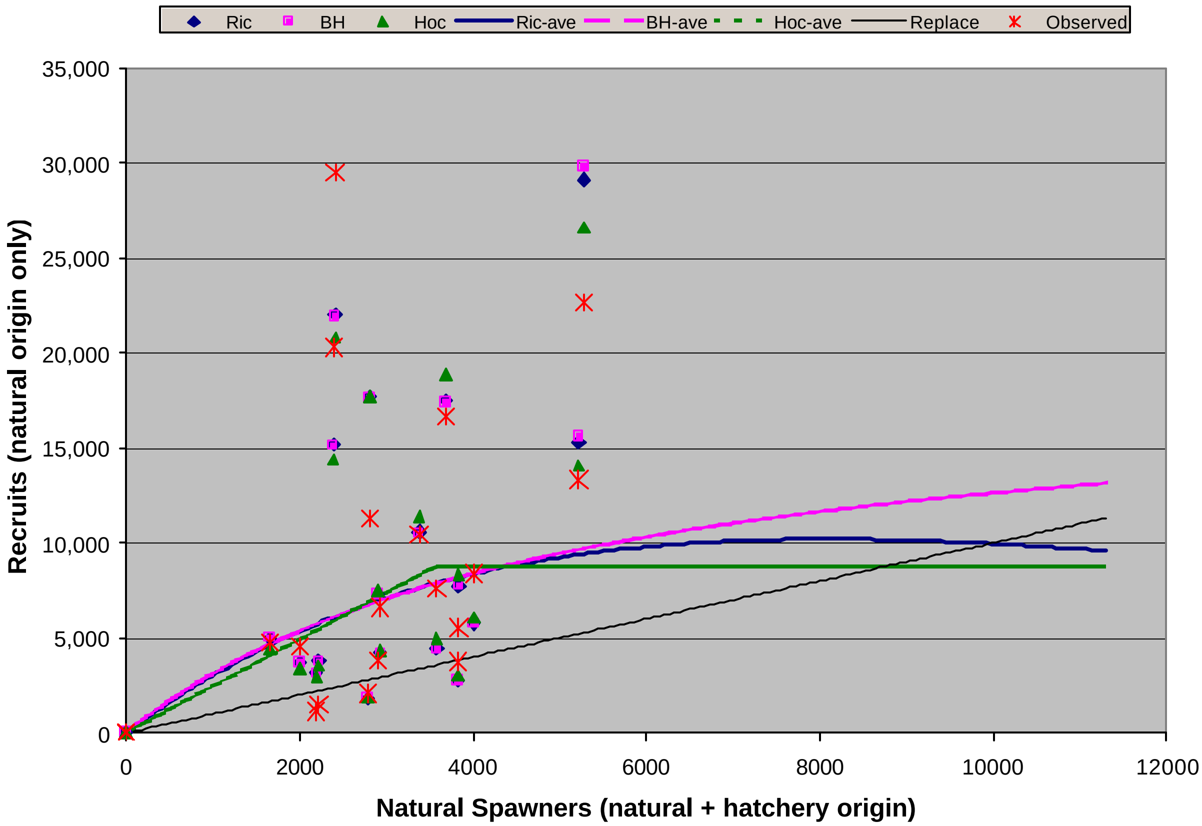
In the above, a is the density independent parameter, b is the density dependent parameter, c is the parameter for marine survival, d is the parameter for the freshwater covariate, M is the index of marine survival, and F is the freshwater correlate, peak Sep-Mar mean daily flow in this case.

Data used for the Skykomish Population

The Skykomish RER was based on analyses of the 1979-1996 brood years. Uncertainty about accuracy of escapement data and completeness of catch data precluded use of data before 1979. The 1996 brood year was the last year for which data were available to conduct a complete cohort reconstruction. There was no evidence of depensation or of a time trend in the data after adjustment for environmental variables.

Results

Figure 1. Comparison of observed and predicted recruitment numbers for the Skykomish chinook



population, brood years 1979 – 1996, under three different models of the spawner-recruit relationship (see text for further details).

The results of model fitting for various combinations of environmental correlates are summarized in Table 7 and graphed in Figure 1. We used the parameters from the fits using the NPS2 marine survival index and using both the marine and freshwater environmental correlates (upper right corner of Table 7).

PROJECTION PHASE

We projected the performance of the Skykomish stock at exploitation rates in the range of 0 to .30 at intervals of .01 using the fitted values of a, b, c, and d for the three spawner-recruit models. All projections were made assuming low marine survival using the average and variance of the marine survival indices observed for the most recent 10-year period. The freshwater environmental correlate (peak winter flow) was projected using the average and variance observed for the entire period used in the model fitting phase. Projections were run for target exploitation rates varying from 0 to .50, in increments of .01. The lower abundance threshold (LAT) was 1,745, derived as described above. The upper abundance threshold was the MSH escapement level (also described above). This biological reference point varies with the assumed marine survival and also with the particular form of the spawner-recruit relationship. We used the average marine survival index for the low marine survival period to obtain the RET for each spawner-recruit function. These values were: 3,500 – Ricker, 3,600 – Beverton-Holt, and 3,600 – hockey stick.

For each combination of spawner-recruit relationship and exploitation rate we ran 1000 25-year projections. Estimated probabilities of exceeding the RET were based on the number of simulations for which the final spawning escapement exceeded the RET. Estimated probabilities of falling below the LAT were based on the number of years (out of the total of 25,000 individual years projected for each combination) that the spawning escapement fell below the LAT. For each spawner-recruit relationship the sequence of Monte Carlo projection running through the exploitation rate range from 0 to .30 started with the same random number seed so that the results for the different spawner-recruit models would be comparable.

Detailed results of these projections are in Table 8, and summarized results are in Table 5. Indicated target exploitation rates are 0.25 – Ricker, 0.27 – Beverton-Holt, and 0.22 – hockey stick. Since there is no basis to choose one of these models over the other, we propose to use the average of these values as the target exploitation rate. This average is 0.24, rounding down to the nearest whole percentage exploitation rate.

Table 5. Results of the VRAP projections of the Skykomish chinook stock under current conditions showing the indicated target exploitation rate for each form of the spawner-recruit relationship.

Model	TgtER	#fish Mort.	% runs extnct	% yrs <LEL	% runs end>UEL	1st Year	LastYrs Ave.
Ricker	0.25	1671	0	4.0	80.0	2123	5711
Bev-Holt	0.27	1889	0	4.5	80.3	2084	6149
H-Stick	0.22	1427	0	3.0	81.3	2172	5747

MANAGEMENT UNIT REBUILDING EXPLOITATION RATE AND LOWER ESCAPEMENT THRESHHOLDS

The management unit maximum exploitation rate was set at 0.24, which is the average of the maximum allowable rates computed for the Skykomish stock using the three different spawner-recruit relationships. This is assumed to provide the appropriate protection to both populations. It was not possible to obtain a fit of the Snoqualmie data to any of the spawner-recruit models, with or without the use of environmental correlates. It is believed that this is due to the fact that some of the escapement estimates for the Snoqualmie are unreliable, and biased low, due to poor visibility in some years.

The lower abundance threshold for management was set starting with critical escapement levels, expands these per population management thresholds, and expands again to a management unit threshold based on the average contribution of each population to the management unit's escapement.

The second step in deriving the management unit lower threshold was to expand each stock's lower management threshold by dividing the percentage of the total escapement that the stock is expected to comprise.

We can then compute the total system escapement required such that we expect each stock to achieve its lower escapement management threshold by dividing the percentage of the total escapement the stock is expected to comprise. The expected percentages of each stock came from the recent 12-year escapement breakout by stock (Table 1). Averaging the ratios of the two stocks' estimated NOR escapements over the twelve years gives an average Snoqualmie fraction of 37.7% of the total.

Table 6. Derivation of the lower management threshold for each Snohomish chinook population and the management unit escapement necessary to achieve this level for each population.

	Snoqualmie	Skykomish
Critical level	400	942
Low R/S	1.01	0.71
Exp. rate	.24	.24
Low threshold	521	1745
Implied MU LT	1,381	2,802

The maximum of the management unit lower thresholds required to achieve the lower thresholds for the two stocks is 2,800 (Table 6), which was chosen as the management unit lower threshold for management planning purposes. Because this is so much higher than the indicated management threshold for protection of Snoqualmie escapement, this Plan is providing extra protection to the Snoqualmie stock pending acquisition of better escapement data.

INTERPRETATION OF FRAM MODEL FOR PRESEASON PLANNING

Currently the comanagers use the Fishery Regulation Assessment Model (FRAM) for preseason planning of total fishery impacts (Table 2). Because a different set of exploitation rates (Table 3) was used in the model fitting phase for Snohomish Chinook, it is important to assess whether preseason exploitation rates from FRAM are directly comparable with the RER derived in the projection phase described above.

The exploitation rates in Tables 2 and 3 cannot be directly compared for a number of reasons. First, the A&P rates (Table 3) are brood year rates, while the FRAM rates (Table 2) are calendar or fishing year rates. FRAM is based on applying current year abundances and fishery exploitation levels to average fishery-specific exploitation rates observed form coded-wire tag recoveries in a base period (Larrie Lavoy, WDFW, personal communication). In contrast the preterminal rates in the A&P tables use current year coded-wire tag recoveries from indicator groups.

Second, FRAM more accurately represents Snohomish Chinook by modeling both the fingerling outmigrant or "ocean type" and yearling outmigrant or "stream type" (Healy 1991) components of the Snohomish run. Comparison of coded-wire tag recoveries from hatchery groups released as age-0 fingerlings as compared with groups released as age-1 yearlings consistently shows differences in patterns of fishery exploitation. FRAM utilizes CWT recovery information from Wallace River (Skykomish) yearling production releases as well as fingerling CWT data to accurately reflect Snohomish Chinook distributions (Larrie LaVoy, WDFW, personal communication). Because yearling recovery data are not incorporated into the A&P tables, these rates may not be an accurate reflection of the true rates for Snohomish Chinook.

Finally, the two models use different set of indicator coded-wire tag groups to represent the Snohomish management unit. This is more difficulty for the Snohomish than for other management units because there is no local indicator coded-wire tag stock available for

Snohomish ocean type Chinook, although a program of double-index tagging at Wallace River hatchery began in 2000 with hopes of developing an appropriate indicator group.

In summary, information available at this time indicates that there is some management risk to using FRAM as we implement annual fishing plans with the intention of achieving our Plan objectives. However, given the uncertainties in estimates associated with estimates of exploitation rates in both the A&P tables and with FRAM, it is not clear that one is more accurate in representing true Snohomish Chinook exploitation rates. Therefore, some additional, precaution is called for in using FRAM to assess whether a given package of proposed fisheries will result in an exploitation rate below the RER guideline of 0.24 for the Snohomish. Therefore, the comanagers will initially use a guideline of 0.21 for the Snohomish instead of the 0.24 derived in the projection phase of this analysis. This guideline was the highest preseason projected exploitation rate for Snohomish since the 2000 application of the comanagers' plan (Table 2). The range of preseason exploitation rates primarily reflects variation in abundance of other chinook stocks and changes in the pattern or level of fisheries outside the comanagers' jurisdiction. Given the procedures in place for annual implementation of the plan, particularly with respect to our intention of not increasing fisheries and our record of managing fisheries to levels that are below exploitation rate ceilings, our expectation is for preseason Snohomish Chinook exploitation rates less than 0.21. Since observed spawning escapements have been increasing during this period (Table 1), consistently above the comanagers' former goal of 5,250 (Ames and Phinney 1977), and generally the largest observed since the beginning of the database in 1965, we feel that recent management has met this plan's objective of reducing fishery impacts so that the population can recover if other factors improve.

In addition, as part of our commitment to evaluate performance of the Plan and modify it as necessary to ensure objectives are achieved, the comanagers intend to l review in detail the implications of the differences between the A&P and FRAM exploitation rates. This may result in the need to recompute RER estimates, compute a quantitative adjustment for FRAM projections.

Data gaps

Priorities for filling data gaps to improve understanding of stock / recruit functions, harvest exploitation rate, and marine survival:

- Annual implementation of a double-index coded-wire tagging program using fingerling summer chinook from Wallace River Hatchery to enable direct assessment of harvest distribution, and estimation of harvest exploitation rates and marine survival rates. (Initiated beginning with the 2000 brood year).

- Estimates of natural-origin smolt abundance from chinook production areas. (Outmigrant trapping began in the Skykomish in 2000 in the Snoqualmie in 2001).

- Estimates of estuarine and early-marine survival for fingerling and yearling smolts.

- Quantification of the contribution of hatchery-origin adults to natural spawning for each stock. (Research is underway. Estimates of hatchery contribution to natural spawning populations is available for the 1997 through 2001 return years.)

Table 7. Results of model fits for different combinations of environmental correlates.

	PS(6) for marine, FW			NPS(2) for marine, FW		
	Ric	Bev	Hoc	Ric	Bev	Hoc
a - productivity	4.1658	0.2400	4.1658	5.1234	0.1782	3.6572
b - Spawners	0.000000	0.000000	42,216	0.000124	0.000035	13,092
c – Marine	0.8330	0.8330	0.8330	0.6418	0.6394	0.6313
d - Freshwater	-0.000011	-0.000011	-0.000011	-0.000014	-0.000014	-0.000014
SSE	2.414	2.414	2.414	0.343	0.345	0.347
MSE (esc)	0.268	0.268	0.268	0.038	0.038	0.039
autocorrelation in error	0.199	0.199	0.199	-0.366	-0.358	-0.449
R	0.680	0.680	0.680	0.895	0.891	0.891
F	2.579	2.579	2.579	12.096	11.569	11.568
PROBABLITIY	0.1184	0.1184	0.1184	0.0016	0.0019	0.0019
MSE (reruits)	0.564	0.564	0.564	0.276	0.278	0.255
autocorrelation in error	-0.390	-0.390	-0.390	-0.133	-0.126	-0.147
Ave.Pred. Error	7237	7237	7237	3994	4092	3999

	No Freshwater, PS(6)			No Freshwater, NPS(2)		
	Ric	Bev	Hoc	Ric	Bev	Hoc
a - productivity	2.8789	0.3474	2.8789	4.6677	0.0761	3.9737
b - Spawners	0.000000	0.000000	42,216	0.000254	0.000132	6,238
c – Marine	0.8398	0.8398	0.8398	0.6986	0.7042	0.7341
d - Freshwater	0.000000	0.000000	0.000000	0.000000	0.000000	0.000000
SSE	2.897	2.897	2.897	1.056	1.057	1.065
MSE (esc)	0.290	0.290	0.290	0.106	0.106	0.106
autocorrelation in error	0.203	0.203	0.203	0.175	0.141	0.116
R	0.617	0.617	0.617	0.862	0.855	0.877
F	3.066	3.066	3.066	14.505	13.605	16.739
PROBABLITIY	0.0915	0.0915	0.0915	0.0011	0.0014	0.0006
MSE (reruits)	0.447	0.447	0.447	0.298	0.304	0.316
autocorrelation in error	-0.372	-0.372	-0.372	-0.071	-0.088	-0.069
Ave.Pred. Error	7773	7773	7773	4310	4437	4089

	No Marine			No Marine or Freshwater		
	Ric	Bev	Hoc	Ric	Bev	Hoc
a - productivity	3.7071	0.2697	3.7071	2.7118	0.3688	2.7118
b - Spawners	0.000000	0.000000	19,851	0.000000	0.000000	66,517
c – Marine	1.0062	1.0000	1.0000	0.5000	0.5000	0.5000
d - Freshwater	-0.000010	-0.000010	-0.000010	-0.000001	-0.000001	-0.000001
SSE	3.463	3.463	3.463	3.758	3.758	3.758
MSE (esc)	0.346	0.346	0.346	0.342	0.342	0.342
autocorrelation in error	0.086	0.086	0.086	-0.017	-0.017	-0.017
R	0.435	0.435	0.435	0.299	0.299	0.299
F	1.164	1.164	1.164	1.076	1.076	1.076
PROBABLITIY	0.3512	0.3512	0.3512	0.3219	0.3219	0.3219
MSE (reruits)	0.768	0.768	0.768	0.789	0.789	0.789
autocorrelation in error	-0.324	-0.324	-0.324	-0.369	-0.369	-0.369
Ave.Pred. Error	7838	7838	7838	7938	7938	7938

Table 8. Summary of projections of the Skykomish population at different target exploitation rates for three different forms of the spawner-recruit relationship.

	Pr(final esc > UAT) %			Pr(ann. Esc. < LAT) %		
Target ER	**B-H**	**Ricker**	**Hockey-St**	**B-H**	**Ricker**	**Hockey-St**
0.00	99.20	96.60	96.30	0.30	0.50	0.50
0.01	99.40	97.80	96.50	0.40	0.70	0.60
0.02	99.00	96.40	95.80	0.50	0.70	0.60
0.03	98.70	95.80	95.60	0.40	0.60	0.50
0.04	98.10	95.60	94.70	0.40	0.70	0.60
0.05	98.40	96.40	95.80	0.50	0.70	0.70
0.06	97.80	95.10	94.30	0.60	0.90	0.80
0.07	97.40	94.70	93.20	0.60	0.90	0.80
0.08	97.80	94.90	94.00	0.60	0.90	0.80
0.09	97.50	94.80	93.70	0.70	1.00	1.00
0.10	97.40	94.20	92.70	0.70	1.00	1.00
0.11	96.90	94.10	92.20	0.90	1.20	1.10
0.12	95.70	92.10	90.50	0.80	1.20	1.20
0.13	96.50	93.40	90.70	1.20	1.60	1.60
0.14	96.00	92.10	90.30	1.10	1.40	1.40
0.15	95.60	90.40	89.30	1.20	1.50	1.60
0.16	93.60	90.90	88.20	1.60	2.00	2.00
0.17	93.70	89.80	87.00	1.50	1.80	2.00
0.18	91.40	87.90	84.60	1.60	1.90	2.10
0.19	91.10	87.70	83.80	2.10	2.50	2.80
0.20	91.00	86.90	83.90	1.90	2.30	2.60
0.21	91.00	87.90	84.40	2.10	2.40	2.80
0.22	90.70	87.30	**82.50**	2.30	2.70	3.00
0.23	86.40	82.70	78.70	2.80	3.20	3.70
0.24	86.40	82.30	77.10	3.40	3.70	4.40
0.25	84.30	**80.00**	75.30	3.50	4.00	4.80
0.26	85.80	82.40	76.90	3.30	3.90	**4.70**
0.27	**80.30**	77.10	71.50	4.50	**4.90**	6.10
0.28	77.90	73.90	68.70	**4.50**	5.00	6.30
0.29	78.40	73.90	65.80	5.10	5.60	7.20
0.30	75.20	72.00	65.60	5.20	5.60	7.50

Lake Washington Management Unit Status Profile

Component Stocks

Cedar River Fall
North Lake Washington Tributaries Fall

Geographic distribution

Fall chinook are produced in three basins in the Lake Washington watershed, the Cedar River, at the south end of Lake Washington; Big Bear Creek and its tributary Cottage Creek (the "Northern Tributaries" which are tributaries of the Sammamish Slough), and Issaquah Creek, the principle inlet at the south end of Lake Sammamish. Historically, chinook also spawned in other smaller tributaries to Lake Washington (e.g. – May and Kelsey creeks) and the Sammamish Slough, (e.g. Little Bear, Swamp, and North creeks). Recent field studies indicate sporadic use of these streams.

About ten miles of Bear Creek, and three miles of Cottage Creek, are accessible to chinook. Recent surveys have located concentrated spawning between RM 4.25 and 8.75 in Bear Creek and the entire three miles of Cottage Lake Creek. Approximately 75% of the total chinook escapement in Bear/Cottage is in Cottage Lake Creek. Spawning in Issaquah Creek occurs predominately in reaches between RM 1 and the Issaquah hatchery (Ames et al. 1975). Chinook surplus to hatchery needs are often passed upstream of the rack and spawn in Issaquah Creek.

In the Cedar River, access above RM 21 has been blocked by the Landsburg diversion dam since its construction in 1901. Access to an additional 15 miles of habitat above Landsburg became available in 2003 with the completion of fish passage facilities. There is very little chinook spawning in the Cedar River downstream of RM 5.0.

Hatchery contribution

Hatchery production currently exists at Issaquah Creek (chinook and coho), the University of Washington (chinook and coho), and the Cedar River (sockeye). Due to present and historic enhancement efforts, adults that return to Issaquah Creek are presumed to be predominately of hatchery origin. Outplants were made to most of the tributaries to the Lake Washington basin from the Issaquah and Green River hatcheries, during the period of record (1952 on). Many of these plants continued through the early 1990s. The one exception is the Cedar River where the last plants were in 1964.

Genetic information

Allozyme analysis of samples collected from Cedar River chinook suggest that this stock is genetically distinct, but closely related to that in the Green River (Marshall, 1995b). Genetic samples from chinook in Bear/Cottage Creek are similar to those from Issaquah Creek. Green River hatchery fish were outplanted into the Cedar River system from 1952 to 1964. Until 1916 the Cedar River drained into the Green River, so a close relationship is not surprising. Sampling and genetic analysis of returns to the North Lake Washington tributaries and other independent tributaries is in progress, and preliminary analysis suggests that chinook in Bear/Cottage Creek have similar genetics to chinook returning to Issaquah Creek.

Life History Traits

Juvenile trapping in the Cedar River has shown that the outmigration is bimodal with most of the fish entering the lake prior to April as fry. A smaller percentage of these fish rear in the river to smolt size and outmigrate between May and July. On the average, 75% of the outmigrants are fry. These fry rear along the lakeshore, growing quickly and leave the lake as zero-age smolts. The smolts that migrate out of the river are thought to reach the Locks about the same time as the fry, although some fish are still migrating out of the river in late July. The migration through the Locks begins in mid-May and continues until at least September. Recent PIT tagging of Cedar River chinook suggests that the Cedar River fish migrate out later in the season than hatchery chinook. The Cedar River chinook fry that rear along the lakeshore are unique in that most, if not all, of the chinook stocks that use a lake for rearing are age one or two smolts. The Lake Washington stocks also have a protracted smolt outmigration, with a large percentage of the run outmigrating after July 1.

Adult chinook enter the Lake Washington basin from late May through September, and enter drainages from mid-August through early November. Spawning is usually complete by mid-November.

Status

Annual monitoring of the return through Ballard Locks has, since 1994, provided in-season assessment of the total abundance of chinook. Escapement surveys are conducted annually on index reaches in the Cedar River (RM 0 – 21.4), Bear Creek (RM 1.3 – 8.8) and Cottage Lake Creek (RM 0 – 2.3), and some of the smaller tributaries to Lake Washington. An additional mile of upper Cottage Lake Creek, above the index reach (i.e. up to RM 3.3), is also routinely surveyed. Hatchery rack counts occur at Issaquah Creek Hatchery and the University of Washington facility. Since 2003, returns of mass marked hatchery releases from Issaquah Creek Hatchery have enabled assessment of natural- and hatchery-origin chinook at the Ballard Locks and in natural spawning escapement.

For Cedar River, the geometric mean escapement (i.e. live fish counts in the index reach) from 1993 – 1997 was 319; for 1998 - 2002 the mean was 327. For the North Lake Tributaries, the 1993 – 1997 mean escapement to index reach (i.e. live count) was 110; for 1998 – 2002 the mean increased to 330 (Table 1).

Table 1. Escapement estimates for of Lake Washington fall chinook, 1993-2002 (MIT et al. 2003), based on live fish counts in the index reaches of the Cedar River (RM 0 – 21.4), and the North Lake Tributaries (RM 1.3 – 8.8 in Bear Creek, and RM 0 – 2.3 in Cottage Lake Creek).

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Cedar River	156	452	681	303	227	432	241	120	810	369
N. Lake Tribs	89	436	249	33	67	265	537	228	458	268

Additional, and more extensive survey coverage and redd counts, conducted since 1999, have improved our understanding of the distribution and abundance of natural spawning for the two Lake Washington populations (Table 2).

Table 2. Redd count-based estimates of escapement to the Cedar River index reach, and live-fish estimates of escapement to upper Cottage Creek (RM 2.3 – 3.3), 1999 – 2002.

	1999	2000	2001	2002
Cedar River – Redd counts	180	53	395	266
- Expanded by 2.5 fish / redd	450	133	988	665
Upper Cottage Creek – live counts	195	104	231	92

Redd count-based estimates for the Cedar River index reach suggest that escapement has substantially exceeded the standard live-count estimates. The supplemental surveys of upper Cottage Lake Creek indicate that approximately 30% of natural spawning in the Bear Creek system has occurred above, and in addition to, that in the index reach. The additional abundance identified in Table 2, when added to the index counts, still does not fully account for escapement to the Cedar River and North Lake tributaries.

Harvest distribution

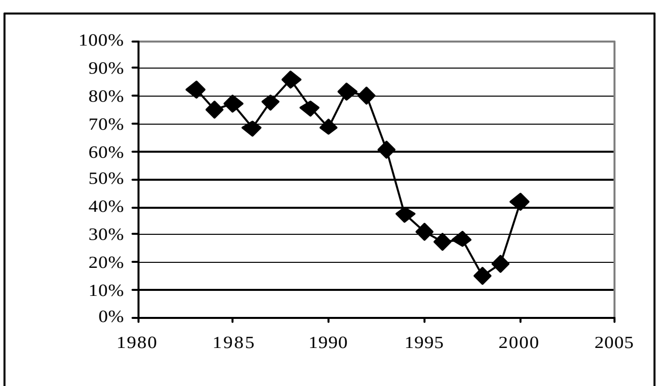
The harvest distribution of Lake Washington chinook has not been directly assessed because representative coded-wire tagged hatchery releases are only available for a few brood years from the Issaquah Hatchery in the late 1980s, and the University of Washington hatchery in the late 90s. However, because of their similar life history and genetic heritage, tagged fingerling releases from Central Puget Sound facilities (Soos Creek hatchery on the Green River, and Grovers Creek Hatchery on the Kitsap Peninsula) facilities provide the best available representation of pre-terminal harvest distribution (see Green River profile).

Terminal harvest of Lake Washington chinook has been minimized since 1994 by regulatory measures that have eliminated directed harvest and reduced incidental impacts in Shilshole Bay, the Ship Canal, and in Lake Washington. Commercial and recreational fisheries directed at sockeye and coho salmon have been specifically shaped to minimize impacts on chinook. Recreational fishing regulations focus effort on Issaquah Hatchery returns.

Exploitation rate trends

Based on post-season FRAM runs, average total annual exploitation rates on the aggregate of natural and hatchery-produced Lake Washington chinook have fallen 66 percent from levels in the 1980s to 1996 – 2000.

Figure 1. Total annual, adult equivalent, fisheries exploitation rate of Lake Washington chinook, estimated by post-season FRAM runs for management years 1983 – 2000.



Management Objectives

The upper management threshold (escapement goal) for the Lake Washington unit is 1,200 (i.e. live count) in the Cedar River index reach. This goal was derived as the average escapement observed from 1965 – 1969, and represents the best available estimate of habitat capacity (Hage et al. 1994). However, current habitat conditions constrain productivity and have prevented achievement of the goal in recent years (Table 1).

The current management objective for the Lake Washington unit is to constrain the exploitation rate, in pre-terminal southern U.S. fisheries, to a level less than or equal to 15%. This objective was derived from highly constrained regimes planned for the 1998 – 2000 management years. Directed terminal fisheries have been closed for ten years, and pre-terminal exploitation rates have been declining. Terminal area fisheries have been reduced to the Minimum Fisheries Regime to conserve Lake Washington chinook, even though forecast abundance has exceed the low abundance threshold. This fishing regime has stabilized escapement.

Management objectives are not currently specified for the North Lake Washington tributaries population. Estimated escapement to the Bear Creek / Cottage Creek index areas averaged 350 during the period from 1983 – 1992 (Hage et al. 1994), and the co-managers previously adopted this as an interim escapement goal. The aforementioned management objectives, for the Cedar River population, provide adequate protection for the North Lake population, as demonstrated by stable escapement levels observed in the last ten years (Tables 1 and 2). The long-term objective for Lake Washington chinook is to increase productivity to the point that the natural escapement goal is regularly met or exceeded.

Anticipating that productivity and abundance will remain low during the term of this plan, the co-managers will continue to implement the recent management actions which constrain impacts on Lake Washington natural chinook to very low incidental levels. These harvest measures ensure that harvest impacts are consistent with recovery of listed stocks. The co-managers will continue to refine their harvest management for Lake Washington natural chinook by shaping terminal fisheries for sockeye and coho to minimize incidental impact on chinook.

The low abundance threshold of 200 for the Cedar River population was set substantially above the historically low escapement from which the stock recovered (e.g. the 1993 escapement of 156). If pre-season fishery simulation modeling indicates that escapement will fall below 200, conservation measures will be implemented to further reduce the pre-terminal SUS exploitation rate to a level no greater than 12%, and terminal fisheries will also be shaped to reduce impacts on Lake Washington chinook, while maintaining fishing opportunity on harvestable sockeye and coho salmon (see Appendix C).

These objectives are intended to maintain the diversity of the naturally reproducing populations that comprise the management unit. Diversity is expressed in various aspects of life history, including the age composition of mature fish, migration timing, and spawning and rearing distribution. Harvest constraint has been exerted, over the last ten years, to maintain stable spawning escapements to the Cedar River and the North Lake tributaries, but is not capable, by itself, of improving their status. If habitat protection and restoration measures succeed in alleviating the primary constraints on productivity in these systems, harvest management will respond by ensuring that spawning escapement is sufficient to optimize production, so that abundance will rebuild.

Data gaps

The highest priority will be placed on collecting the data needed to quantify the productivity of Lake Washington stocks. Until the fundamental aspects of productivity are defined it will be difficult to assess the success of recovery actions, whether they entail improvement in habitat productivity or production supplementation.

Table 3. Data gaps related to harvest management, and projects required to address those data needs.

Data gap	Research needed
Estimates of total spawning escapement for each stock.	Mark/recapture study, repeated for a minimum of three years; or an alternate approach to expanding index reach counts to total escapement. First done in FY2000
Estimates of natural smolt production in Issaquah Creek.	Fry/smolt trapping in Issaquah Creek to supplement ongoing trapping in the Northern Tributaries and the Cedar River.
Quantification of fry and smolt survival in Lake Washington and the Ship Canal.	Smolt trapping at the locks to quantify mortality as smolts transit the lake and the locks. Trapping at the locks has proven to be very difficult.
Quantification of freshwater predation on smolts	Continuation of the Lake Washington Studies Project to further quantify fish, bird and lamprey predation. Fish predation research has been completed and is being written up. Bird predation work has not been started
Comprehensive estimates of incidental fishing mortality.	Creel surveys of recreational fisheries that target other species. The approach should be research oriented.
Estimates of bias in ladder counts at Ballard Locks, relative to spawning ground surveys.	Tagging and tracking of adult chinook from the locks and the ladder to estimate repeat passage. Started in 1998, research is complete and is awaiting write-up.
Estimate of spawning and production above Landsburg Dam	Spawner surveys to account for fish passed above the dam, fry/smolt trapping at or near the dam to independently assess upper basin productivity and survival.
Estimates of hatchery stray rates for Cedar and North Lake Tributaries	All ages are ad-clipped beginning in 2004. Enumerate ad-clipped fish during spawner surveys; sample for and collect CWTs.
Assess pre-spawning mortality	Quantify pre-spawning mortality related to environmental variables like water temperature.

Related Data Questions

Is chinook survival from emergent fry to adult (smolt?) correlated with early life history strategy? (i.e. – what are the relative survival rates of fry outmigrants compared to smolt outmigrants in the Cedar River). Is survival different in the upper basin than it is in the lower basin?

Is scour of chinook redds related to the magnitude of peak flow events in the Cedar River, and the position of redds in the stream channel?

What is the relationship between flow at Landsburg and the availability of water at the Locks for operating the smolt slides?

Green River Management Unit Status Profile

Component Stocks

Green River Fall Chinook

Geographic description of spawner distribution

Fall chinook are produced in the mainstem Green River and in two major tributaries - Soos Creek and Newaukum Creek. Adults that spawn in Soos Creek are presumed to be predominantly of hatchery origin. However, recent investigations into straying raise questions regarding this, and other assumptions related to run reconstruction. (See stock status, below). Newaukum Creek spawners appear to be closely related to the spawners in the mainstem.

Spawning in the mainstem Green River occurs from RM 26.7 up to RM 61. Spawning access higher in the drainage is blocked by the City of Tacoma's diversion dam, and at RM 64 by Howard Hanson Dam. Spawning occurs in the lower 10 miles of Newaukum Creek. Adults returning to the hatchery at RM 0.7 of Soos Creek may also spawn naturally and adults surplus to program needs at the Soos Cr. Hatchery are often passed upstream.

Life History Traits

Fall chinook begin entering the Green River in July, and spawn from mid-September through October. Ocean-type freshwater life history typifies summer/fall stocks from South Puget Sound, with 99 percent of the smolts outmigrating in their first year (WDFW 1995 cited in Myers et al 1998). A long-term average of the age composition of adults returning to the Green River indicates the predominance of age-4 fish (62 percent), with age-3 and age-5 fish comprising 26 percent and 11 percent, respectively (WDF et al 1993, WDFW 1995 cited in Myers et al 1998).

Status

The SASSI review (WDF et al 1993) classified Green River chinook as healthy, because spawning escapement had consistently met the objective since 1978. Spawning escapement has increased recently, with the mean of the 1997–2002 escapement (9077) exceeding that for the preceding five-year period (4799). Total escapement fell below the nominal goal of 5,800 in 1992 – 1994, which triggered an assessment of factors contributing to the escapement shortfall by the PFMC (PSSSRB 1997). However, escapement has exceeded the goal in each subsequent year.

Table 1. Spawning escapement of Green River Fall Chinook, 1992-2002.

1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
5,267	2,476	4,078	7,939	6,026	9,967	7,300	9,100	6170	7975	13950

It is known that returns from hatchery production contribute substantially to natural spawning in the Green River and tributaries. Viability of the naturally spawning stock, absent the hatchery contribution, is uncertain because hatchery returns may be masking poor natural productivity (Myers et al 1998). Analysis of coded wire tags recovered from the spawning grounds and the in-river fishery has yielded highly variable results. Collection of data from Chinook mass-marked since 2000 began in 2003 and is expected to provide better estimates of straying and contribution as analysis is completed.

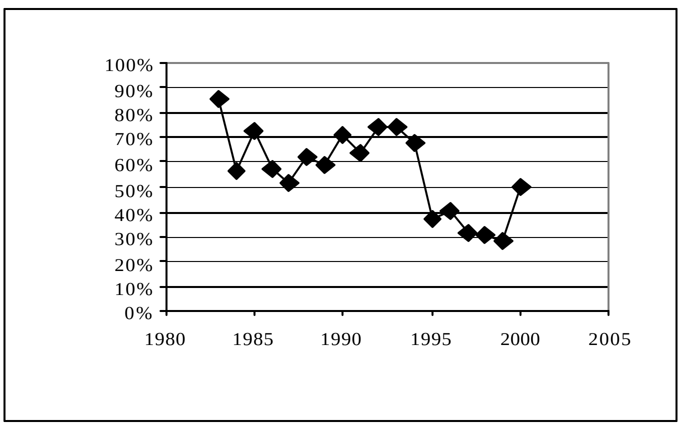
The nominal escapement goal is based on approximate estimates of escapement in the 1970's, and may not reflect the productivity constraints associated with current degraded habitat, but will be used to guide fisheries management until natural capacity is better quantified. Escapement estimation methods are under review. Surveys have been expanded in recent years to calibrate assumptions regarding the relationship between index area counts and total escapement and the third year of a mark/recapture method, also for the purpose of calibration of escapement estimates, was just completed.

Hatchery facilities currently operate on Soos Creek, Keta Creek and Icy Creek. Broodstock has always been collected from local returns, so the hatchery stock presumably retains its native genetic character. Allozyme analysis has shown no detectable difference between hatchery-reared and naturally spawning adults (Marshall et al 1995).

Harvest distribution and exploitation rate trends:

Post-season FRAM runs, incorporating actual catch and stock abundance indicate that annual exploitation rates for Green River chinook have declined 45 percent from levels in the 1980s to 1996 – 2000 (Figure 1). As noted above, recent years' spawning escapement has consistently exceeded the goal.

Figure 1. Total annual, adult equivalent, fishery exploitation rates for Green River chinook for management years 1983 – 2000, estimated by post-season FRAM runs.



Coded-wire tagged fingerling releases from the Green River (and Grovers Creek) describe harvest distribution in recent years. Fisheries in British Columbia and Alaska account for 32 percent of total fishing mortality. Washington recreational and Puget Sound net fisheries account for 38 percent and 24 percent of total mortality, respectively (Table 3).

Table 3. The harvest distribution of Green River chinook, expressed as a proportion of total annual, adult equivalent exploitation. (CTC 2003).

	Alaska	B.C.	Washington Troll	Puget Sound net	Washington sport
1997 – 2001	2.1%	30.1%	9.4%	23.7%	37.7%

Management Objectives

The co-managers manage fisheries to meet or exceed the spawning escapement goal of 5,800 Green River chinook. This goal has been met or exceeded in 10 of the last 13 years. The co-managers expect that the goal will continue to be met or exceeded as a result of this management approach. The co-managers expect to further refine their management plan for Green River chinook in response to on-going ESA recovery planning, to ensure harvest impacts are consistent with recovery of listed stocks and emerging policies for hatchery management. When the escapement is expected to be less than 5,800, the co-managers will discuss what additional actions, beyond those identified below, may be appropriate to bring the escapement above the 5,800 level.

Management objectives for Green River chinook include an exploitation rate objective for pre-terminal Southern U. S. fisheries and a procedure to manage terminal-area fisheries that is based on an inseason abundance triggers to assure that the escapement goal will be achieved. This management regime assures that harvest of Green River chinook will not impede recovery of the ESU.

Washington preterminal fisheries impacts on Green River chinook are managed at or below a 15 percent 'SUS' exploitation rate, as estimated by the FRAM model. Pre-terminal fisheries include the coastal troll and recreational fisheries managed under the Pacific Fisheries Management Council, and commercial net and recreational fisheries in Puget Sound outside of Elliott Bay.

Due to more restrictive pre-terminal fisheries in recent years, a greater proportion of allowable harvest has been available in the terminal fishery in Elliott Bay and the Duwamish (lower Green) River, where tribal net fisheries and recreational fisheries are managed on the basis of terminal abundance triggers.

Terminal area abundance is estimated annually utilizing a test fishery conducted since 1989. Using this data, two thresholds (triggers) have been set below which planned directed fisheries would not proceed. A value below 100 chinook for the test fishery would cause cancellation of subsequent commercial and sport fisheries. A value below 1000 chinook for the first commercial opening would cause cancellation of any further chinook-directed fishing. These values corresponded with a total run of about 15,000 chinook.

Management thresholds were met in 2000, 2001, 2002 and 2003. Terminal area chinook-directed treaty net and sport fisheries were implemented as scheduled. Natural escapement for 2000, 2001 and 2002 are provided in Table 1. The preliminary estimate for 2003 escapement is more than 7000 spawners.

A critical-abundance threshold of 1,800 natural spawners is established for the Green River management unit on the basis of the lowest observed escapement resulting in a higher escapement four years later. If natural escapement is projected to fall below this threshold during pre-season planning, then additional management measures will be implemented in accordance with procedures established in Appendix C, to minimize fishery-related mortalities.

Data gaps

Several aspects of the productivity of Green River chinook are potentially affected by hatchery-origin fish spawning naturally. The abundance, timing, spawning distribution, and age structure of natural-origin chinook may be masked by the presence of hatchery-origin fish. The viability of the natural origin population cannot be accurately assessed without determining the effects of hatchery straying, so the need for this information will prioritize research. Below are descriptions of the data needs and how they are being addressed.

Data need	Related project
Quantification of the proportion of natural escapement that is comprised of hatchery strays.	Completion of a CWT data set for refinement of current CWT-based estimates. (work in progress) Mass marking of hatchery production. (Brood years 1999-2002 marked
Re-evaluation of escapement estimation methodology	Expanded surveys to calibrate expansion of index area data to total. (begun in 1998 – work continues.) Mark/recapture study to independently calibrate total escapement estimate in association with expanded survey effort. (done in 2000-2002, report in progress)
Estimation of the number of Chinook fry and smolts that emigrate annually from the mainstem Green and Newaukum Creek.	Trap placement in the mainstem Green 1999-2002)
Estimation of differential survival of natural and hatchery origin Chinook in-situ in the Green.	A literature review of methodologies that may have utility for an in-situ experiment should be done.
Estimation of estuarine hooking mortality if selective fisheries are proposed for Elliott Bay.	A literature review and preliminary study design should be done.

White River Spring Chinook Management Unit Profile

Component stocks

White River Spring Chinook

Geographic description

White River Spring Chinook are trapped at the Puget Sound Energy diversion dam in Buckley and transported into the upper watershed, above Mud Mountain Dam, where they spawn primarily in the West Fork White River, Clearwater River, Greenwater River, and Huckleberry Creek. They also spawn in the lower mainstem White, below the diversion dam at RM 23.4 where river conditions preclude estimates of spawner abundance.

The White River population is the only spring stock still present in southern Puget Sound, is geographically isolated from summer/fall stocks, and genetically distinct from all other chinook stocks in Puget Sound. The White River Hatchery program, and the Minter Hupp Complex supplement production. The stock has, in past years, been maintained as captive brood at the Hupp Springs and Peale Pass net pen facilities. The supplementation program is considered essential to recovery, so hatchery production is included in the listed ESU.

Life History Traits

Spring chinook enter the Puyallup River from May through mid-September, and spawn from mid-September through October. All adipose-bearing fish arriving at the Buckley trap without detectable CWT's are passed upstream. CWT fish are transferred to the White River Hatchery and confirmed as White River Spring Chinook by genetic testing before they are incorporated into the broodstock supplementation program.

Fry emerge from the gravel in late winter and early spring. In contrast to other spring stocks in Puget Sound, White River chinook smolts emigrate primarily (80 percent) as subyearlings (SSSCTC 1996), after a short rearing period of three to eight weeks. Adults mature primarily at age-3 or age-4.

Status

Escapement of White River chinook exceeded 5,000 in the early 1940's, but the construction of hydroelectric and flood control dams, and degradation of the spawning and rearing habitat, reduced abundance to critical levels in the 1970's. Escapement was less than 100 through the 1980s and fell below 10 in 1984 and 1986. A supplementation program has been operating since 1971, and it has succeeded in raising escapement to levels between 300 and 600 in recent years (Table 1). The geometric mean of escapement in 1992 – 1996 was 477, and for the three more recent years, 413.

Table 1. Spawning escapement of White River spring chinook, 1993-2002.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Upper River	409	392	605	630	400	316	553	1523	2002	803
Broodstock	1444	2033	1982	924	822	454	429	740	814	
Total	1853	2425	2587	1554	1222	770	982	2263	2816	

The upper river figure represents untagged fish captured at the Buckley trap and transported to upstream spawning grounds (ACOE data cited in HGMP). Broodstock includes collections at Minter Creek, South Sound Net Pens, and the White River Hatchery, and excludes jacks through 1995 (WDFW et al. 1996 cited in HGMP). Broodstock values from 1996 on represent collection at White River Hatchery only.

The status of White River spring chinook has been considered critical. Returns in recent years have improved, but evaluation of natural-origin versus hatchery-origin returns is not complete. Degraded spawning and rearing habitat, and the migration blockage imposed by dams, currently imposes severe constraints on natural productivity. The contribution of natural-origin adults to spawning escapement has not been quantified, but there is evidence to suggest that the stock is not currently viable in the absence of supplementation. The supplementation program succeeded in raising escapement above the critically low levels seen in the 1970's and 1980s, and it may continue to protect the viability of the stock, but natural production will not recover until the habitat constraints are addressed.

Harvest distribution and exploitation rate trends

Based on recoveries of coded-wire tagged yearling released from White River and Hupp Springs hatcheries during calendar years 1996 – 2000, 90 percent of the total harvest mortality of White River springs has taken place in Puget Sound recreational fisheries. An average of five percent of total mortality occurred in British Columbia fisheries.

Table 2. The recent average distribution of annual harvest mortality for yearling White River spring chinook, expressed as a proportion of total annual adult equivalent exploitation rates (CTC 2003)

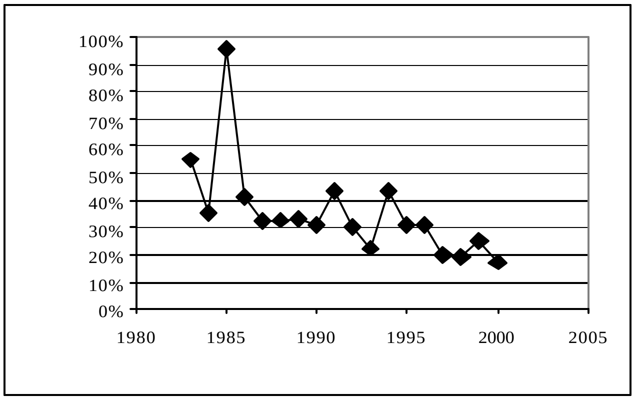
	Alaska	B.C.	Wa troll	PS net	Wa sport
1996 - 00	0.0%	5.4%	0.8%	3.9%	90.0%

Increasingly conservative management of Washington fisheries has resulted in a declining trend in total exploitation rate over the last six years, as estimated by post-season FRAM runs that incorporate actual catch and stock abundance (Figure 1). The average rate for management years 1998 – 2000 was 61 percent lower than the average for management years 1983 – 1987. . The fisheries simulation model (FRAM) has been modified to incorporate only White River fingerling tag codes, which show a slightly different harvest distribution than yearlings that comprise the PSC Indicator Stock.

Management Objectives

Fisheries in Washington will be managed to achieve a total exploitation rate, including fisheries in British Columbia, no greater than 20 percent. This exploitation rate ceiling, which is three points higher than the ceiling in the 2001 Harvest Management Plan, reflects changes in coded-wire tag and historical catch data incorporated in the most recent calibration of FRAM (L. LaVoy, WDFW, memorandum to co-manager technical staff, February 12, 2002). Achievement of this rate requires continued constraint of Puget Sound net and recreational fisheries, and allows minimal tribal ceremonial and subsistence fisheries in the river. Tag recovery and escapement data are insufficient, at present, to support direct assessment of the productivity of the stock.

Figure 1. Total annual, adult equivalent fisheries exploitation rate for White River Spring Chinook for management years 1983 – 2000, estimated by post-season FRAM runs.



The current management objective constrains fishing mortality and, in recent years, has provided spawning escapement well in excess of the critical threshold of 200. Escapement below this level is believed to present significant risk to genetic diversity and exposure to depensatory mortality factors, particularly when considering the low productivity of naturally spawning fish.

If preseason fishery simulation modeling suggests that escapement will not exceed the low abundance threshold, further conservation measures will be implemented in fisheries that catch White River chinook, so as to reduce their total exploitation rate to a level that is defined by modeling the fishing regime described in Appendix C. A conservative approach is warranted in managing this stock, and projected escapement near the critical threshold, or failure to achieve broodstock collection objectives, will be considered grounds to re-institute the captive brood program.

Data gaps

- Description of spawning distribution in the upper White River system.
- Quantification of hatchery- and natural-origin adults on the spawning grounds.
- Estimation of natural smolt production.
- Estimation of pre-spawning mortality of adults that are trapped and transported above Mud Mountain dam.

Puyallup River Fall Chinook Management Unit Status Profile

Component Stocks

Puyallup River fall chinook
South Prairie Creek fall chinook

Geographic description

Fall chinook spawn primarily in South Prairie Creek (a tributary of the Carbon River) up to RM 15, the Puyallup mainstem up to Electron Dam at RM 41.7 , the lower Carbon River up to RM 8.5, Voights's Creek, Fennel Creek, Canyon Falls Creek, Clarks Creek, Clear Creek and Kapowsin Creek, and, possibly, the lower White River. Surplus Voights Creek Hatchery adult chinook are currently released to spawn naturally above the Electron diversion and juvenile chinook produced at the Puyallup Voights Creek Hatchery are outplanted to acclimation ponds in the upper Puyallup River, above the diversion dam. Construction of a fishway at Electron Dam is expected to re-establish adult access to the upper river, however, downstream juvenile passage is still deficient in the near future.

Life History Traits

Hatchery programs have introduced non-native stocks, primarily of Green River origin, into the Puyallup system, so it is not clear that naturally spawning chinook bear the native genetic legacy. A remnant native stock may persist in South Prairie Creek, though genetic testing to date has not been conclusive in that respect.

Freshwater entry into the Puyallup River begins in late July, and spawning occurs from mid-September through mid-November. Based on scale samples collected in 1992-93, returning adults were primarily (76 percent) age-4, and age-3 and age-5 fish made up 16 and 6 percent of the sample (WDF et al. 1993 cited in Myers et al. 1998). South Prairie Creek age samples taken between 1992 and 2002 provides a mean age composition, based on brood contribution of the 1991-1997 broods, of 1.0% age-2, 19.1% age-3, 67.3% age-4, 12.3% age-5 and 0.3% age-6 fish (WDFW, unpublished data). Juveniles exhibit ocean-type life history, primarily, with estimated 97 percent of smolts emigrating as subyearlings (WDF et al. 1993 cited in Myers et al. 1998).

Status

Between 1994 and 2001, escapement to the South Prairie Creek sub-basin has ranged from 667 to 1430 fish, averaging 1048. The turbid nature of the Puyallup and Carbon rivers, due to its their glacial origin, makes enumeration of spawners or redds difficult in the mainstem, so the accuracy of the system-wide estimates is uncertain.

The former nominal escapement goal, that was intended principally to assure adequate broodstock to hatchery programs, was 3,250, including natural spawning and escapement to the hatcheries.

Harvest distribution and exploitation rate trends:

The harvest distribution of Puyallup fall chinook has not been assessed, because a local indicator stock has not been consistently coded-wire tagged. Distribution in pre-terminal fisheries is likely similar to that of the South Sound fingerling indicator stock, which is composed of tagged releases from the Green River (Soos Creek) and Grovers Creek. This distribution is shown, above, in the Green River profile.

Post-season FRAM runs, which incorporate actual catch in all fisheries and actual abundance of all chinook stocks, indicate the total, annual, adult-equivalent exploitation rate for Puyallup fall chinook declined sharply from 1995 – 1998, and that rates have since increased as improved survival has enabled increased harvest, while still achieving the escapement objectives.

Management Objectives

Since the existence of an indigenous fall chinook stock in the Puyallup system is uncertain, and current natural production is substantially augmented by hatchery-origin fish, the harvest management objectives will reflect the need to adequately seed natural spawning areas until the productive capacity of habitat is quantified, and the existence of an indigenous stock is resolved. Until recently fisheries were managed to supply adequate broodstock to the hatchery programs.

The harvest management objective for Puyallup fall chinook is to not exceed a total exploitation rate of 50 percent, to assure that a viable, natural-spawning population is perpetuated. Pre-season fisheries planning, to not exceed this ceiling rate, has been shown to result in spawning escapement of more than 500 to the South Prairie Creek - Wilkeson Creek complex. . Though escapement estimation methods have evolved recently to better quantify total fall chinook escapement to the entire Puyallup system, as previous described, water clarity in South Prairie Creek still affords the most reliable index.. Achieving escapement to South Prairie / Wilkeson of at least 500, according to the most recent surveys, indicates that the entire system is seeded adequately to assure viable natural production. Based on more comprehensive spawning surveys, including monitoring of recolonization of the basin above Electron Dam, the co-managers expect, in the near future, to develop a system escapement goal for fall chinook.

Pre-terminal and terminal fisheries in Puget Sound were constrained in 1999 and 2000 to achieve this objective. The productive capacity of habitat in South Prairie Creek, or in the Puyallup mainstem and tributaries is not quantified, so a system-wide escapement goal has not been established. By reducing the total exploitation rate, relative to those levels in the early- to mid-1990s, this harvest regime will is intended to provide stable or increasing levels of natural escapement. Achieving higher natural escapement, under the new management objective, will experimentally probe the productivity of natural spawners in the system.

A low abundance threshold of 500 spawners, for the entire system, is established for the Puyallup fall management unit. If escapement is projected to fall below this threshold, fisheries-related mortality will be reduced to a level defined by the fisheries regime described in Appendix C. The threshold is set above the point of stock instability, to prevent escapement from falling to that level which incurs substantial risk to genetic integrity, or expose the stocks to depensatory mortality factors.

Should the forecast, terminal-area abundance of Puyallup chinook fall below the low abundance threshold, and the forecast be confirmed by the evaluation fishery in the river (see below), extraordinary conservation measures would be implemented to limit harvest mortality and provide for natural spawning escapement. Directed chinook fishing (i.e., during the fall chinook management period) would be reduced to no more than one day per week for tribal fishers to meet their ceremonial and subsistence needs. Recreational fisheries would be limited to mark selective fisheries in the Carbon River. With concomitant reductions in preterminal fishing mortality, the total SUS exploitation rate would be expected to be approximately 25%.

Data gaps

- Improve spawning escapement estimates for the Puyallup River and/or validate the use of South Prairie Creek and Wilkeson Creek counts as an index for the system.

- Estimate the contribution of hatchery- and natural-origin adults to natural spawning , by mass-marking hatchery production. Brood year 1999 hatchery production was 100% marked.

- Develop a spawner – recruit function for natural-origin, naturally spawning chinook to validate the recovery exploitation rate objective. This task is dependent on completion of the two preceding tasks.

- Conduct an evaluation fishery, during the early weeks of the fall chinook management period, in the Puyallup mainstem, to collect catch and catch-per-effort data that may, in future, become the basis for in-season assessment of stock abundance. Statistical models relating catch or CPUE to abundance will, in addition to several other sources of information regarding migration timing and progress of the river fishery, inform the fishery managers regarding possible changes in the fishery schedule, should these indicators suggest that abundance differs significantly from the pre-season forecast.

Nisqually River Chinook Management Unit Status Profile

Component Stocks
Nisqually fall

Geographic description

Adult chinook ascend the mainstem of the Nisqually River to river mile 40, where further access is blocked by the La Grande and Alder dams, facilities that were constructed for hydroelectric power generation by the City of Tacoma's public utility. It is unlikely that chinook utilized higher reaches in the system, prior to the dams' construction. Below La Grande dam the river flows to the northwest across a broad and flat valley floor, characterized by mixed coniferous and deciduous forest and cleared agricultural land. Between river miles 5.5 and 11 the river runs through the Nisqually Indian Reservation, and between river miles 11 and 19 through largely undeveloped Fort Lewis military reservation. At river mile 26, a portion of the flow is diverted into the Yelm Power Canal, which carries the water 14 miles downstream to a powerhouse, where the flow returns to the mainstem at river mile 12. A fish ladder provides passage over the diversion. Both Tacoma's and Centralia's FERC license requires minimum flows in the mainstem Nisqually.

Fall chinook spawn in the mainstem above river mile 3, in numerous side channels, as well as in the lower reaches of Yelm Creek, Ohop Creek, the Mashel River and several smaller tributaries. Production is augmented by production at the Kalama Creek and Clear Creek hatcheries, which are operated by the Nisqually Tribe.

Life History Traits

Adult fall chinook enter the Nisqually River system from July through September, and spawning activity continues through November. After emerging from the gravel, juveniles typically spend two to six months in freshwater before beginning their seaward migration. Residence time in their natal streams may be quite short, as the fry usually move downstream into higher order tributaries or the mainstem to rear. Extended freshwater rearing for a year or more, that typifies some Puget Sound summer/fall chinook stocks, has not been observed in the Nisqually system.

Returning adults mature primarily at age-3 and age-4, comprising 45 and 31 percent, respectively (WDF et al. 1993, WDFW 1995 cited in Myers et al. 1998).

Stock Status

It is generally agreed that native spring and fall chinook stocks have been extirpated from the Nisqually River system, primarily as a result of blocked passage at the Centralia diversion, de-watering of mainstem spawning areas by hydroelectric operations, a toxic copper ore spill associated with a railroad trestle failure, and other freshwater and marine habitat degradation (Barr, 1999). Studies are underway to determine whether any genetic evidence suggests persistence of the native stock. Initial results indicate that the existing naturally-spawning and hatchery stocks are identical, and were derived from hatchery production that utilized, principally, Puyallup River and Green River fall chinook. Like other stocks in South Puget Sound, in which current production is based on naturalized and supplemented returns from a hatchery program, the Nisqually has been managed to achieve escapement sufficient to provide broodstock to the enhancement program.

Natural escapement has met the escapement goal of 1,100 since 1999. The escapement intent shifted and the goal was increased to 1,100 for the 2000 management year (see below). Recent natural spawning escapement has ranged from 340 to 1,700 (Table 2), and hatchery returns have ranged from 1370 to 13,481, in the period between 1993 and 2002. Escapement surveys are difficult in the mainstem river because of the turbidity caused by glacial flour.

Table 1. The abundance of fall chinook returning to the Nisqually River system.

	River Net	Escapement		
Year	Catch	Hatchery	Natural	Total
1993	4024	1370	1655	3025
1994	6183	2104	1730	3834
1995	7171	3623	817	4440
1996	5365	2701	606	3307
1997	4309	3251	340	3591
1998	7990	4067	834	4901
1999	14614	13481	1399	14880
2000	6836	4923	1253	6176
2001	14098	7612	1079	8691
2002	11687	10794	1532	12326

Harvest distribution and exploitation rate trend:

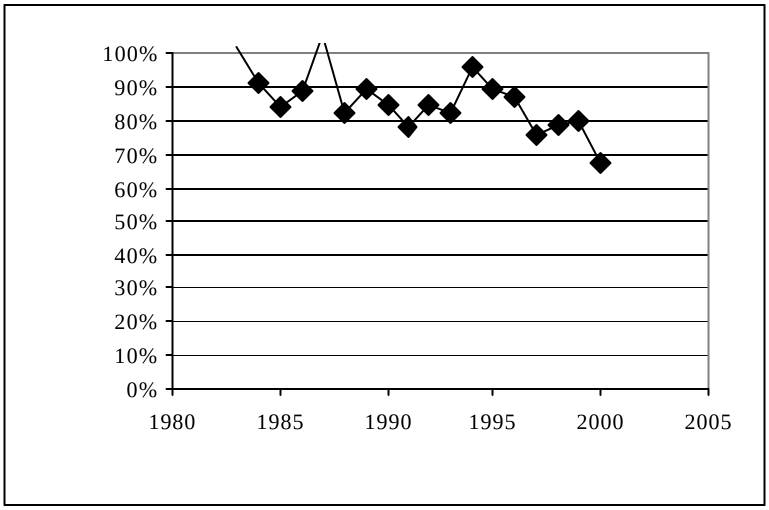
The harvest distribution of Nisqually chinook has been described by analysis of coded-wire tagged fingerling chinook released from Clear Creek and Kalama Creek hatcheries. In recent years 15 percent of the total harvest mortality has occurred in British Columbia and Alaska, primarily in Georgia Strait. Washington troll fisheries have accounted for 14 percent of total fishery mortality. Recreational (ocean and Puget Sound) and net fisheries in Puget Sound , have accounted for 43 and 39 percent of total mortality, respectively.

Table 2. The recent average harvest distribution of Nisqually River fall chinook, expressed as the proportion of annual, adult equivalent fisheries exploitation rate (CTC 2003)
.

	Alaska	B.C.	Washington Troll	Puget Sound net	Washington sport
1997 – 2001	0.5%	14.2%	3.5%	38.7%	43.1%

The total annual exploitation rate for Nisqually chinook has declined slightly since 1993, as described by post-season FRAM runs (Figure 1). FRAM rates are assumed to accurately index the recent trend in exploitation rate, but may not accurately quantify annual exploitation rates, because of the lack of CWT data in the model base period,

Figure 1. Total annual, adult equivalent fisheries exploitation rate of Nisqually fall chinook, from 1983 – 2000, estimated by post-season FRAM runs.



Management Objectives

Because the Nisqually management unit is not a unique, native stock, the need to optimize natural production from natural-origin spawners will be balanced with the fishery enhancement objectives of the hatchery programs. In this sense, the Nisqually unit is similar to other South Puget Sound and Hood Canal natural units where production comprises non-native, introduced chinook stocks, and where natural productivity is severely constrained by habitat degradation. For these units, management intent is distinct from other Puget Sound management units in which production comprises, primarily, native, naturally-spawning stocks.

Analysis of habitat capacity, using the Ecosystems Diagnosis and Treatment methodology (NCRT 2001), enabled derivation of a Beverton-Holt spawner – recruit function that expresses the production potential for a sequence of life stage segments in the mainstem river and major tributaries under currently existing habitat conditions (Moussali and Hilborn 1986). Solution of this production function by standard methods (Hilborn and Walters 1992) estimated that optimum productivity (MSY) under current habitat conditions is achieved by escapement of 1100.

A rebuilding exploitation rate has not been developed for the Nisqually chinook stock. Further analysis, enabled by better quantification of natural escapement, and assessment of the contribution of natural-origin adults to that escapement, mayl allow development of a rebuilding exploitation rate harvest objective based on natural productivity.

The terminal fisheries are managed based on an inseason runsize estimated by the relationship of total runsize and catch success for the tribal commercial net fishery. This method for updating the runsize in-season will initially be applied with information through the third week of August. Subsequent updates will be conducted as catch data continues to accumulate. To enable the fishery to be managed for the 1,100 escapement goal, managers will translate the total runsize to an expected escapement by making an assumption of the proportion of the total run that will spawn naturally. When the in-season update indicates that the escapement goal (1,100) will not be achieved, terminal area fisheries will be constrained by agreement between the co-managers with the objective of increasing spawner abundance to a level at or above the escapement goal.

If forecasted abundance declines very dramatically from the levels observed in recent years, and the in-season assessment confirms the forecast, the comanagers will implement extraordinary conservation measures for the terminal commercial and recreational fisheries to insure the viability of the population. Such measures may include reduced fishing schedules prior to and after the update at the end of August, and closure of chinook-directed fishing in September, after the update. The subsequent coho fishery may be shaped to reduce incidental chinook mortality, but opportunity to catch the entire harvestable surplus of coho will be maintained. In any case, limited chinook harvest will occur as necessary to meet the ceremonial and subsistence needs of tribal members.

Data gaps

- Improve total natural escapement estimates, including age-specific estimates of both natural and hatchery-origin recruits and develop stock-recruit analysis.
- Test the accuracy of the in-season assessment of extreme terminal abundance, and improve the in-season update model as new data allows.
- Quantify the current natural productivity of the system.

Skokomish River Management Unit Status Profile

Component Stocks

Skokomish summer/fall

Geographic description

Spawning takes place in the mainstem Skokomish River up to the confluence with the South and North forks, in the South Fork of the Skokomish River, primarily below RM 5.0, and in the North Fork up to RM 17, where Cushman Dam blocks higher access. Most spawning in the North Fork occurs below RM 13, because flow fluctuation associated with operations of the hydroelectric facility limit access and spawning success higher in the system (WDF et al. 1993).

On the North Fork Skokomish, two hydroelectric dams block passage to the upper watershed. However, a small, self-sustaining population of landlocked chinook salmon is present in Lake Cushman, upstream of the dams. Adults spawn upstream of the lake in the North Fork Skokomish River from river mile 28.2 to 29.9 during November.

Life History Traits

Genetic characterization of the Skokomish chinook stocks has, to date, been limited to comparison of adults and juveniles collected from the Skokomish River with adults from other Hood Canal and Puget Sound populations. Genetic collections were made during 1998 and 1999 in the Skokomish River and there appeared to be no significant genetic differentiation between natural spawners and the local hatchery population. It appears that Hood Canal area populations may have formed a group differentiated from south Puget Sound populations, possibly indicating that some level of adaptation may be occurring following the cessation of transfers from south Sound hatcheries (Anne Marshall, WDFW memo dated May 31, 2000). Current adult returns are a composite of natural- and hatchery-origin fish. During 1998 and 1999, known hatchery-origin fish comprised from 13% to 41% of the samples collected on the natural spawning grounds. Genetic analysis of samples collected from Lake Cushman was inconclusive as to stock origin, and the adults sampled exhibited low genetic variability. (Marshall, 1995a).

Summer/fall chinook enter the Skokomish River starting in late July with the majority of the run entering from mid-August to mid-September. Chinook in the Skokomish River spawn from mid-September through October with peak spawning during mid-October. Adults mature primarily at age-3 (33%) and age-4 (43%); the incidence of age 2 fish (jacks) is highly variable. In 1999, based on a sample of 143 fish, the age composition of naturally-spawning chinook in the Skokomish River system was estimated to be 2.8% age 2, 58.0% age 3, 38.5% age 4, and 0.7% age 5 fish (Thom H. Johnson, WDFW memo dated November 8, 2000). In 2000 and 2001, the age composition of naturally spawning chinook was 16.1% and 1.2% age 2, 11.3% and 58.3% age 3, 71.0% and 36.9% age 4, and 1.6% and 3.6% age 5, respectively (Thom H. Johnson, pers. Comm.. 12/3/02). Consistent with most other summer/fall populations in Puget Sound, naturally produced smolts emigrate primarily during their first year; 2 percent of the smolts may migrate as yearlings (Williams et al. 1975 cited in Myers et al. 1998). In the Skokomish River, most naturally-produced chinook juveniles emigrate during the spring and early summer of their first year of life as fingerlings (Lestelle and Weller 1994).

Status

The SASSI classified Hood Canal summer/fall chinook as a single stock of mixed origin (both native and non-native) with composite production (sustained by wild and artificial production) (WDFW et al. 1992). The combination of recent low abundances (in all tributaries except the Skokomish River) and widespread use of hatchery stocks (often originating from sources outside Hood Canal) led to the conclusion in SASSI that there were no remaining genetically unique, indigenous populations of chinook in Hood Canal. However, a sampling effort is currently under way (led by WDFW in cooperation with NMFS and Treaty Tribes) to collect genetic information from chinook juveniles and adults in the tributaries of Hood Canal. This investigation is intended to provide further information on the genetic source and status of existing chinook populations.

The existence of historical, indigenous populations, that have not been significantly impacted by past management practices and that have remained distinct and sustainable is at least questionable. The genetic sampling effort referenced above is intended to help resolve remaining uncertainty about the existence of any historical, indigenous populations. In the interim, management measures have been formulated to provide reasonable protection for naturally spawning chinook and adequate flexibility for future change.

Historically, the Skokomish River supported the largest natural chinook production of any stream in Hood Canal. However, habitat degradation has severely reduced the productive capacity of the mainstem and South Fork portions of the system. As previously noted, the North Fork has been blocked by two hydroelectric dams. Hatchery chinook production has been developed at Washington State's George Adams and McKernan hatcheries to augment harvest opportunities and to provide partial mitigation for reduced natural production in the Skokomish system, primarily caused by the North Fork dams. The Skokomish Tribe, whose reservation is located near the mouth of the river, has a reserved treaty right to harvest chinook salmon.

Over the period from 1998 – 2002, natural spawning escapement ranged from 926 to 1,913, exceeding the nominal goal of 1,650 twice (Table 1)

Table 1. Total spawning escapement of Skokomish River fall chinook, 1993 - 2002.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Hatchery	612	495	5196	3100	1885	5584	8227	4033	8816	8828
Natural	960	657	1398	995	452	1177	1692	926	1913	1,479
Total	1572	1152	6594	4095	2337	6761	9919	4959	10729	10307

Harvest distribution and exploitation rate trends:

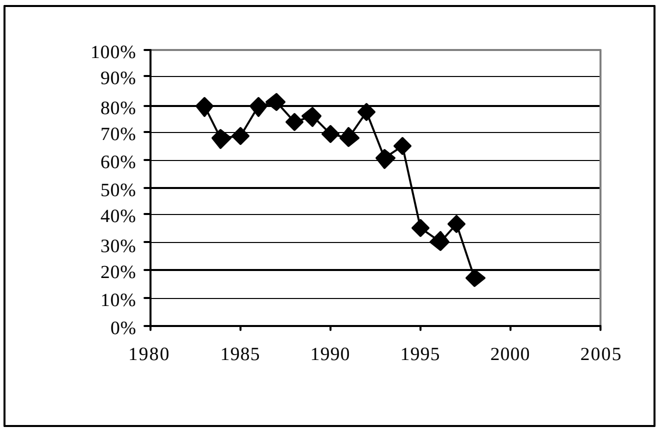
The harvest distribution of Skokomish chinook is best described by recovery of coded-wire tagged fingerlings released from George Adams Hatchery. The average for calendar years 1996 – 2000 indicates that 33 percent of harvest mortality was associated with Canadian and Alaskan fisheries, 13 percent with Washington ocean troll fisheries, 48 percent in recreational fisheries, and 10 percent with net fisheries in Puget Sound.

Table 2. Average harvest distribution of Skokomish River summer/fall chinook, for management years 1997 – 2001, as percent of total adult equivalent fishery mortality (CTC2003).

Years	Alaska	B.C.	Washington troll	Puget Sound net	Washington sport
1997-2001	2.4%	30.9%	8.9%	10.2%	47.7%

The total annual (i.e., management year) exploitation rate, computed by post-season FRAM runs, declined substantially between 1991 and 1998 (Figure 1). The subsequent increase in exploitation rate reflects increased abundance, due in part to improved marine survival, which has allowed higher harvest while still meeting escapement objectives.

Figure 1. Total fishery-related, spawner equivalent exploitation rates of Skokomish River summer/fall chinook for management years 1983 – 1998, estimated by post-season FRAM runs.



Management Objectives

The immediate and short-term objective for Skokomish River is to manage chinook salmon as a composite population (including naturally and artificially produced chinook). The composite population will be managed, in part, to achieve a suitable level of natural escapement; and to continue hatchery mitigation of the effects of habitat loss; and to provide to the Skokomish Tribe partial mitigation for its lost treaty fishing opportunity. Habitat recovery and protection measures will be sought to improve natural production. Over time, alternative management strategies will be explored that may lead to improved sustainable natural production, and reduced reliance on mitigative hatchery support for the Skokomish stock and fisheries.

The nominal escapement goal for the Skokomish River is 3,650. It is the sum of spawner requirements for 1,650 in-stream spawners (HCSMP; 1985) and 2,000 spawners required for the maintenance of on-station hatchery production (see 1996 Production Evaluation MOU, PNPTC-WDFW-USFWS; 2002 Framework Plan, WDFW-PNPTT). Recent composite escapements have been substantially above the 3,650 fish level, averaging 6,941 for the 1997 – 2001 period, and exceeding the 3,650 goal in four of the last five years. In the same period, natural escapement has averaged 1,332, and exceeded 1,650 twice. Escapements to the hatchery have averaged 5,709 fish and have exceeded the 2,000 fish goal in four of the last five years. (Table 1).

The escapement goal of 3,650, along with its component requirements for natural and hatchery spawners, (WDF Tech. Rept. 29, 1977; PSSMP, 1985; HCSMP, 1985; HCSMP Prod MOU,

1996) is intended to maintain full hatchery mitigation and meet current estimates of MSY escapement to natural spawning areas, under current habitat conditions.

A low abundance threshold escapement of 1,300, represents the aggregate of 800 natural spawners and 500 adults returning to the hatchery rack. At these levels, the hatchery escapement component represents the minimum requirement to maintain production. The natural escapement component threshold is set at approximately 50% of the current MSY estimate and represents a level necessary to ensure in-system diversity and spatial distribution (Magnuson-Stevens Act, National Standard for Overfishing Review Threshold). In the 1997 – 2001 period, the critical threshold was exceeded in all years for this management unit. Component critical thresholds in these years were exceeded in all years for hatchery escapement, and in four of the last five years for natural escapement.

During the recovery period, pre-terminal fisheries in southern U.S. areas (SUS), will be managed to ensure a ceiling rate of exploitation of 15%, or less, as estimated by the FRAM model (est. of 1997-1999 SUS preseason impacts). Pre-terminal fisheries include the coastal troll and recreational fisheries managed under the Pacific Fisheries Management Council, and commercial and recreational fisheries in Puget Sound, outside Hood Canal. Terminal fisheries are managed to achieve the escapement goal of 3,650. If the recruit abundance is insufficient for the goal to be met, OR regardless of the total escapement, the naturally spawning component of this population is expected to fall below 1,200 spawners, OR the hatchery component is expected to result in less than 1,000 spawners, additional terminal fishery management measures will be taken, with the objective of meeting or exceeding these spawner levels. The following management measures have been taken in recent years for this purpose, and will be considered in 2003:

- Commercial and recreational fisheries in northern Hood Canal areas (WDFW Areas 12 and 12B) will be reduced or eliminated in the months of July through September.
- Commercial and recreational fisheries in southern Hood Canal areas (WDFW Areas 12C and 12D) will be "shaped" to direct the majority of the fishing effort to the Hoodsport Hatchery zone, thus greatly reducing impacts to the Skokomish Management Unit. In 2000, approximately 90% of the total commercial harvest in Area 12C was directed at, and taken, in that zone.
- In the Skokomish River, Treaty Indian commercial fisheries will be limited in August and September, to areas upstream of the Skokomish delta milling area (upstream of the SR 106 crossing), and downstream of the U.S. 101 crossing.
- In the Skokomish River, recreational salmon fisheries will be limited, through September, to areas upstream of the mouth and downstream of the U.S. 101 crossing.

If, despite the implementation of the above measures, the projected escapement is expected to be less than 1,300 total spawners, OR regardless of the total escapement, the naturally spawning component of this population is expected to fall below the critical threshold of 800 spawners, OR the hatchery component is expected to result in less than 500 spawners, pre-terminal SUS fisheries will be constrained to minimize mortality, in accordance with conservation measures described in Appendix C, or more restrictive measures that have been evaluated and agreed-to by the co-managers for the year in question. In Hood Canal terminal areas, additional management measures will be taken, with the objective of meeting or exceeding these critical spawner levels.

All of the measures shall initially be based on preseason forecasted abundance and escapement projections and may be adjusted during the season, following any inseason reassessment of the terminal abundance. As of 2002, the Co-managers have investigated the feasibility of developing a sufficiently accurate method to derive in-season estimates of abundance, using available commercial and/or recreational, as well as hatchery and/or natural escapement data. However, no approach was found that would result in better estimates when compared to preseason forecasts.

This management regime recognizes the need to optimize natural production in the Skokomish River. However, production potential is currently severely constrained by reduced habitat capacity and quality in the South Fork, and by the influence of the hydroelectric and re-regulation dams on the North Fork. The current productive capacity of habitat has not been quantified in terms of the number of adults required to fully seed the available spawning area or optimize smolt yield.

Principles that underlie the current management intent for Skokomish River chinook include:

Full recovery of natural productivity in the Skokomish River cannot occur under the current hydroelectric operating regime and degraded habitat status;

The management regime will provide adequate seeding of existing habitat and insure the maintenance of in-system diversity and spatial distribution by assuring that (if available) at least 800, and up to 1,650 (the currently estimated level of MSY), natural spawners reach the spawning grounds;

Natural production is dependent on the mitigative hatchery program to partly support natural escapement;

Hatchery- and natural-origin spawners appear to be genetically similar, and have demonstrated their capacity to adapt to the Skokomish River environment.

Access to harvest opportunity on returning adults produced by the enhancement program at George Adams Hatchery is mandated as partial mitigation for the effects of operation of the City of Tacoma's hydroelectric facility.

The recovery objective for the ESU, which includes conservation and rebuilding of natural production that is representative of the geographic and genetic diversity that characterizes the ESU, is served, in part, by assuring that natural production of locally-adapted populations is recovered in the mid-Hood Canal streams (Duckabush River, Dosewallips River, and Hamma Hamma River) where habitat quality does not constrain to the extent that it does in the Skokomish River.

Management objectives for the Skokomish River management unit will evolve in response to improved understanding of natural productivity, and success in restoring the productive potential of habitat in the system.

Data gaps

- Continue to improve escapement estimates for the South and North Forks of the Skokomish River.

- Develop means to assess the contribution of Skokomish hatchery and natural origin adults to the fishery and to hatchery and natural escapements.

- Quantify the current natural productivity (in terms of recruits per spawners) and natural capacity (in terms of adults and juvenile migrants) of the system.

Mid-Hood Canal Management Unit Status Profile

Component Sub-populations

Hamma Hamma River summer/fall
Dosewallips River summer/fall
Duckabush River summer/fall

Geographic description

Chinook spawn in the Hamma Hamma River mainstem up to RM 2.5, where a barrier falls prevents higher access. Spawning can occur also in John Creek when flow permits access. A series of falls and cascades, which may be passable in some years, block access to the upper Duckabush River at RM 7, and to the upper Dosewallips River at RM 14. Spawning may also occur in Rocky Brook Creek, a tributary to the Dosewallips. Most tributaries to these three rivers are inaccessible, high gradient streams, so the mainstem provides nearly the entire production potential.

Life History Traits

Genetic characterization of the mid-Hood Canal Management Unit (MU) has, to date, been limited to comparison of adults returning to the Hamma Hamma River in 1999 with other Hood Canal and Puget Sound populations. These studies, although not conclusive, suggest that returns to the Hamma Hamma River are not genetically distinct from the Skokomish River returns, or recent George Adams and Hoodsport hatchery broodstock (A. Marshall, WDFW unpublished data). The reasons for this similarity are unclear, but straying of chinook that originate from streams further south in Hood Canal, and hatchery stocking, could be contributing causes.

Status

The Mid-Hood Canal MU is comprised of chinook local sub-populations in the Dosewallips, Duckabush and Hamma Hamma watersheds. These sub-populations are at low abundance (Table 1). Current chinook spawner surveys are typically limited to the lower reaches of each stream. In the Hamma Hamma River, the majority of the chinook spawning habitat is currently being surveyed. In the Dosewallips and Duckabush rivers, however, the areas surveyed are transit areas and do not include all spawning areas. Upper reaches of the Dosewallips and Duckabush have been more routinely surveyed since 1998, but few chinook adults or redds have been observed. Prior to 1986 no reliable estimates are available because all escapement estimates for these rivers were made by extrapolation from the Skokomish River.

Table 1. Natural spawning escapement of Mid-Hood Canal fall chinook salmon, 1993-2002.

River	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
HammaHamma	28	78	25	11	na	172	557	381	248	32
Duckabush	17	9	2	13		57	151	28	29	20
Dosewallips	67	297	76	na		58	54	29	45	43
Total	142	384	103	na		287	762	438	322	95

In 1992, SASSI classified Hood Canal summer/fall chinook as a single stock of mixed origin (both native and non-native) with composite production (sustained by wild and artificial production) (WDFW et al. 1992). The combination of recent low abundances (in all tributaries except the Skokomish River) and widespread use of hatchery stocks (often originating from sources outside Hood Canal) led to the conclusion in SASSI that there were no remaining genetically unique, indigenous populations of chinook in Hood Canal. A study is currently underway to characterize the genetic profile of chinook juveniles and adults in the mid-Hood Canal MU.

In 2002, when SASSI was updated to SaSI, mid-Hood Canal chinook were classified as a single stock, comprised of chinook salmon which currently spawn in the Hamma Hamma, Duckabush and Dosewallips watersheds (WDFW et al. 2002). In 2002, the stock status was rated as "Critical" in SaSI, primarily because of chronically low spawning escapements whose average escapement abundance, over the 1991 – 2002 period, failed to meet the established low escapement threshold of 400.

Harvest distribution and exploitation rate trends:

The harvest distribution of mid-Hood Canal chinook, and recent fishery exploitation rates, cannot be directly assessed because none of the component sub-populations have been coded-wire tagged. However, it is reasonable to assume, given their similar life history, that tagged fingerling chinook released from the George Adams Hatchery, on the Skokomish River, follow a similar migratory pathway and experience mortality in a similar set of pre-terminal fisheries in British Columbia and Washington. A summary of recent analyses of the Skokomish River data are shown in that profile.

Management of the terminal area fisheries in Hood Canal enables some separation of harvest between Skokomish/ Hoodsport and the mid-Hood Canal natural MU. With only Hoodsport and Skokomish tags available to model terminal impacts, the selective intent of the terminal regime will be estimated based on the freshwater entry period for mid-Canal rivers, and the distribution of historical net catch among the sub-areas of Hood Canal.

It is reasonable to conclude that mid-Hood Canal sub-populations experienced a decline similar to that of Skokomish River chinook, but their total exploitation rate has been lower, because the terminal area fishery, which can harvest a significant proportion of Skokomish chinook, has been restricted to the southern end of Hood Canal since the early 1990s.

Management Objectives

The management objective for the mid-Hood Canal MU is to maintain and restore sustainable, locally adapted, natural-origin chinook sub-populations. Management efforts will initially focus on increasing the abundance in the MU and its local, natural sub-populations. Fisheries are being restricted to accommodate the escapement objectives.

The existence of historical, indigenous populations that have remained distinct and sustainable is at least questionable and while additional genetic sampling may help resolve any remaining uncertainty, the Co-managers' intent is to support their ongoing local diversity adaptation.

During the recovery period, fisheries in southern U.S. areas (SUS), will be managed to achieve a preterminal (PT) AEQ rate of exploitation of less than15%, as estimated by the FRAM model (see Section IV). This exploitation rate is the same as that for the remainder of the Hood Canal management units because no means exist to separately assess the exploitation of the mid-Hood Canal unit, and there is no indication that its exploitation pattern is different between Hood Canal

MUs. In this case, preterminal fisheries include the coastal troll and recreational fisheries managed under the Pacific Fisheries Management Council, and the marine commercial and recreational fisheries in Puget Sound. The extreme terminal areas for this management unit include the freshwater areas in each river.

The migratory pathway and harvest distribution of mid-Hood Canal chinook is presumed to be similar to that of the Skokomish River indicator stock, although that stock's return continues past the mid-Canal area and reaches the Skokomish River, farther south. The FRAM simulation model suggests that the terminal (Area 12C) and extreme-terminal (in-river) fisheries may harvest up to 25% of the Skokomish terminal run. However, terminal-area fisheries at the far southern end of Hood Canal, near the mouth of or in the Skokomish River, are not believed to harvest significant numbers of adults returning to the mid-Hood Canal rivers of origin. Time and area restrictions are believed to be effective in relieving harvest pressure on the mid-Hood Canal sub-populations.

When the escapement goal of 750 spawners (established as interim MSY in Hood Canal Salmon Management Plan (HCSMP)) is not expected to be met, recreational and commercial fisheries will be adjusted to the extent necessary to exert a PT SUS AEQ exploitation rate of less than 15%, or meet the escapement target, whichever occurs first. These measures shall also include the closure of all extreme terminal (freshwater) fisheries that are likely to impact adult spawners of these sub-populations. These measures will be considered in order to ensure that the PT SUS AEQ exploitation rate will not exceed 15%.

A low abundance threshold of 400 chinook spawners has been established for the mid-Hood Canal MU, which is approximately 50% of the current MSY goal for the mid-Hood Canal sub-populations, in the HCSMP (1985). If escapement is projected to fall below this threshold, further conservation measures will be implemented in pre-terminal and terminal fisheries to reduce mortality and ensure that the projected PT SUS AEQ exploitation rate does not exceed 12.0%. The best available information indicates that escapement has been below the low abundance threshold in three out of the last five years. The co-managers recognize the need to provide across-the-board conservation measures in this circumstance, and to avoid an undue burden of conservation falling on the terminal fisheries.

Unless genetic studies conclude that distinct populations persist in individual mid-Hood Canal streams, the primary focus of management will be to ensure that sufficient spawners escape to these systems to maintain self-sustaining sub-populations. These sub-populations will contribute geographic diversity to the ESU by their adaptation to the unique environmental conditions found in these drainages of the east slope of the Olympic Mountains.

Data gaps

- Continue to improve escapement estimates
- Test the accuracy of the pre-season forecasts
- Develop means to assess the origin composition of adults in the escapement
- For each sub-population, and the MU, reassess spawner requirements and quantify the current productivity (in terms of recruits per spawner) and capacity (in terms of adults and juvenile migrants).

Dungeness Management Unit Status Profile

Component Stocks

Dungeness River chinook

Distribution and Life History Characteristics

Chinook spawn in the Dungeness River up to RM 18.9, where falls, just above the mouth of Gold Creek, block further access. Spawning distribution , in recent years, has been weighted toward the lower half of the accessible reach with approximately two-thirds of the redds located downstream of RM 10.8. Chinook also spawn in the Graywolf River up to RM 5.1.

The entry timing of mature chinook into the Dungeness River is not described precisely, because of chronically low returns of adults. It may occur from spring through September. Adult weir operations in 1997 and 2001 indicate that most of the adult chinook return has entered the river by early August. Spawning occurs from August through mid-October (WDF et al. 1993). At the current low level of abundance, no distinct spring or summer populations are distinguishable in the return. Chinook typically spawn two weeks earlier in the upper mainstem than in the lower mainstem (WDF et al. 1993). Ocean- and stream-type life histories have been observed among juvenile chinook in the system, with extended freshwater rearing more typical of the earlier-timed segment (Ames et al. 1975). Hirschi and Reed (1998) found that a significant number of chinook juveniles overwinter in the Dungeness River.

Smolts from the Dungeness River exhibit primarily an ocean-type life history, with age-0 emigrants comprising 95 to 98 percent of the total (WDF et al. 1993, Smith and Sele 1995, and WDFW 1995 cited in Myers et al. 1998). Adults mature primarily at age four (63%), with age 3 and age 5 adults comprising 10% and 25%, of the annual returns, respectively (PNPTC 1995 and WDFW 1995 cited in Myers et al. 1998).

Stock Status

The SASSI report (WDF et al. 1993) classified the Dungeness spring/summer as critical due to a chronically low spawning escapement to levels, such that the viability of the stock was in doubt and the risk of extinction was considered to be high. Dungeness chinook continued to be classified as critical in the SaSI report (WDFW 2003) because of continuing chronically low spawning escapements.

The nominal escapement goal for the Dungeness River is 925 spawners, based on historical escapements observed in the 1970's and estimated production capacity re-assessed in the 1990s (Smith and Sele 1994). This goal has not been achieved in the past 17 years. The mean spawning escapement level, since 1998, has been 298 (Table 1). It should be noted however that the increase in escapements, observed in recent years, is partly due to a captive brood supplementation program.

Table 1. Spawning escapement of Dungeness River chinook 1986 - 2002.

Return Year	Escapement
1986	238
1987	100
1988	335
1989	88
1990	310
1991	163
1992	153
1993	43
1994	65
1995	163
1996	183
1997	50
1998	110
1999	75
2000	218
2001	453
2002	633
1998 – 2002 Mean: 298	

Chinook production in the Dungeness River is constrained, primarily, by degraded spawning and rearing habitat in the lower mainstem. Significant channel modification has contributed to substrate instability in spawning areas, and has reduced and isolated side channel rearing areas. Water withdrawals for irrigation during the migration and spawning season have also limited access to suitable spawning areas.

The co-managers, in cooperation with federal agencies and private-sector conservation groups, have implemented a captive brood stock program to rehabilitate chinook runs in the Dungeness River. The primary goal of this program is to increase the number of fish spawning naturally in the river, while maintaining the genetic characteristics of the existing stock. The first returns of age-4 adults, from the brood year 1996 release of 1.8 million fingerlings, occurred in 2000. Uncertainty over the survival of these fingerlings has led managers to project abundance conservatively, (i.e., discount the potential return from supplementation).

In addition to the broodstock program, the local watershed council (Dungeness River Management Team) and a work group of state, tribal, county and federal biologists have been working on several habitat restoration efforts. Based on the 1997 report, "Recommended Restoration Projects for the Dungeness River" by the Dungeness River Restoration Work Group, local cooperators have installed several engineered log jams, and acquired small riparian refugia properties. Other projects including larger scale riparian land acquisition, dike setback, bridge lengthening and setback, as well as estuary restoration are in the planning, analysis and proposal phases.

Management Objectives

The management objective for Dungeness chinook is to stabilize escapement and recruitment, as well as to restore the natural-origin recruit population basis through supplementation and fishery restrictions. Pre-terminal incidental harvest is constrained to a ceiling AEQ exploitation rate of

10.0% in the southern U.S. Directed terminal commercial and recreational harvests have not occurred in recent years, and incidental harvest in fisheries directed at coho and pink salmon have been regulated to limit chinook mortality .

Direct quantification of the productivity of Dungeness chinook will require either the accumulation of sufficient coded-wire tag recoveries to reconstruct cohort abundance, or an alternate method of measuring freshwater (egg-to-smolt) and marine survival. Releases from the supplementation program are represented by coded-wire tagged groups, adipose fin marked groups, otolith marked groups and blank wire tag groups. Recoveries of these tags, otoliths, and marks will enable cohort reconstruction. However, given the degraded condition of spawning and rearing habitat in the lower mainstem, it must be assumed that current natural productivity is critically low. The captive brood supplementation program will be suspended, following production from the 2003 brood year.

The lack of stock specific historical tag information has necessitated the interim use of a neighboring representative stock in fishery simulation modeling of Dungeness chinook salmon. Tagged Elwha Hatchery fingerlings are used by the FRAM to estimate the harvest distribution and exploitation rates for all Strait of Juan de Fuca chinook management units. (See Elwha Profile, below). Also, for units with very low abundance, such as the Dungeness, the FRAM model's accuracy may be limited. However, the co-managers will continue to develop and adopt conservation measures that protect critical management units, while realizing the constraints on quantifying their effects in the simulation model.

Lacking sufficient direct assessment of the productivity of Dungeness chinook, it may be appropriate to examine what is known about other Puget Sound management units with similar life history and similar status. The status of Nooksack River early chinook, in particular the South Fork Nooksack management unit, is also classified as critical, due to chronically low spawning escapement. Degraded habitat is known to constrain freshwater survival in the Nooksack system, as it does in the Dungeness. The recovery exploitation rate of the Nooksack units has been estimated to be 20 percent (NMFS 2000). The harvest objective for Dungeness (i.e., to maintain exploitation in southern U.S. fisheries below 10 percent), implies a total exploitation rate of 20 percent or less, given that approximately half of the harvest of Dungeness chinook may occur in southern fisheries.

The critical escapement threshold for the Dungeness River is 500 natural spawners, which is approximately 50% of the escapement goal. Whenever natural spawning escapement for this stock is projected to be below this threshold, SUS fisheries will be managed to further reduce incidental mortality. Until the supplementation program is successful in rebuilding returns to levels sufficient to provide escapement levels above this threshold, harvest will be constrained, to SUS incidental AEQ impacts of less than 6.0%.

Data gaps

- Describe freshwater entry timing
- Continue to collect scale or otolith samples to describe the age composition of the terminal run.
- Describe the fishery contribution and estimate fishery-specific exploitation rates from CWT recoveries.
- Estimate marine survival.
- Estimate annual smolt production per spawner (i.e. , freshwater survival)

Elwha River Management Unit Status Profile

Component Stocks

Elwha River chinook

Geographic Distribution and Life History Characteristics

Summer chinook spawn naturally in the portions of the lower 4.9 miles of the Elwha River, below the lower Elwha dam, though most of the suitable spawning habitat is below the City of Port Angeles' water diversion dam at RM 3.4. Their productive capacity is very low, because of extremely restricted suitable habitat. Their productivity is also very low due to severely altered and degraded spawning and rearing habitat, and high water temperatures during the adult entry and spawning season, which contribute to pre-spawning mortality (see Table 2, below).

Entry into the Elwha River begins in early June and continues through early September. Spawning begins in late August, and peaks in late September and early October (WDF et al. 1993). Elwha chinook mature primarily at age 4 (57%), with age 3 and age 5 fish comprising 13% and 29%, of annual returns, respectively (WDF et al. 1993, WDFW 1995, PNPTC 1995 cited in Myers et al. 1998).

Naturally produced smolts emigrate primarily as subyearlings. Roni (1992) reported that 45 to 83% of Elwha River smolts emigrated as yearlings, and 17 to 55 percent as subyearlings, but this study did not differentiate naturally produced smolts from hatchery releases of yearlings. The Elwha Channel facility no longer releases yearling smolts.

Status

Elwha River chinook were designated as "healthy" in the SASSI document (WDF et al. 1993), which considered productivity in the context of the currently available habitat for natural production. However, in the past decade, the total spawner goal of 2,900 was not met in any year (see Table 1). Therefore, in the SaSI report (WDFW 2003), the Elwha Management Unit was classified as depressed, because of the negative escapement trend and chronically low levels of spawning escapement. The stock is a composite of natural and hatchery production. In the Elwha River, chinook production is limited by two hydroelectric dams which block access to upstream spawning and rearing habitat. Recovery of the stock is dependent on removal of the two dams, and restoration of access to high quality habitat in the upper Elwha basin and certain tributaries. Chinook produced by the hatchery mitigation program in the Elwha system are considered essential to the recovery, and are included in the listed ESU.

The comanagers have concluded that recovery of the Elwha stock is not possible unless the dams are removed and access to pristine, productive habitat, which lies largely within Olympic National Park, is restored.

The nominal spawning escapement goal of 2,900 for Elwha River chinook has not been achieved, even in the absence of in-river fishery impacts, in the past 10 years. The average number of spawners over the last five years has been 2,079, which is somewhat higher than the average of the preceding five years (1993-1997), which was 1,611..

Table 1. Total spawning escapement of Elwha River chinook, 1993 – 2002.

1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
1,562	1,216	1,150	1,608	2,517	2,358	1,602	1,851	2,208	2,376

Pre-spawning mortality has been a significant factor affecting natural and hatchery production in the Elwha system. High water temperature during the period of freshwater entry and spawning is exacerbated by impoundment of the river behind the two upstream dams. It contributes directly to prespawning mortality, and in some years, promotes the infestation of adult chinook by *Dermocystidium*. Pre-spawning mortality has ranged up to 68% of the extreme terminal abundance (Table 2), largely due to parasitic infestation.

Table 2. Prespawning mortality of Elwha River chinook.

Return Year	Hatchery Voluntary Escapement	In-River Gross Escapement	Gaff-Seine Removals	Hatchery Prespawn Mortality	In-River Prespawn Mortality	Total Prespawn Mortality
1986	1,285	1,842	505	376	482	27.4%
1987	1,283	4,610	1,138	432	1,830	38.4%
1988	2,089	5,784	506	428	50	6.1%
1989	1,135	4,352	905	148	412	10.2%
1990	586	2,594	886	160	64	7.0%
1991	970	2,499	857	108	N/A	3.1%
1992	97	3,762	672	26	2,611	68.3%
1993	165	1,404	771	7	0	0.5%
1994	365	1,181	749	61	269	21.3%
1995	145	1,667	518	37	625	36.5%
1996	214	1,661	1,177	147	120	14.2%
1997	318	2,209	624	3	7	0.4%
1998	138	2,271	1,551	51	0	2.1%
1999	113	1,512	609	23	0	1.4%
2000	177	1,736	1,021	62	0	3.2%
2001	195	2,051	1,396	38	0	1.7%
2002	473	1,943	1,080	40	0	1.7%

Harvest Distribution and Exploitation Rate Trend

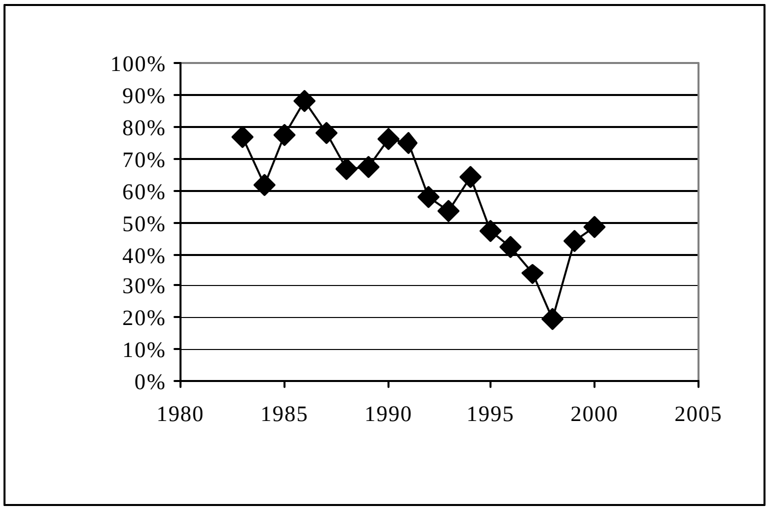
Based on recoveries in 1993 – 1997 of tagged fingerlings released from the local hatchery, Elwha River chinook are a far-north migrating stock, as evidenced by 16% and 59% of total mortality occurring in Alaskan and British Columbian fisheries, respectively (Table 3). Net fisheries in Puget Sound account for only 1% of total fishing mortality, and Washington troll and sport fisheries account for 11%, and 22%, respectively.

Table 3. The average distribution of adult equivalent annual fishing mortality for Elwha River chinook, estimated from post-season FRAM runs (CTC 2003)

Years	Alaska	B.C.	Wash. Troll	Puget Sound Net	Washington sport
1993 – 97	16.2%	58.8%	1.9%	0.8%	22.3%

Post-season FRAM simulations indicate that the total exploitation rate of Elwha River chinook has exhibited a declining trend since 1988 (Figure 1). These post-season FRAM estimates represent the aggregate of JDF units, but are believed to correctly represent the trend in ER for the Elwha unit. The 1998 – 2000 mean exploitation is 51% lower than the average from the 1983 – 1987 period.

Figure 1. Total adult-equivalent exploitation rate for Elwha River chinook, estimated by post-season FRAM runs.



Management Objectives

Fisheries in Washington waters, including those under jurisdiction of the Pacific Fisheries Management Council, when the escapement goal is not projected to be met, will be managed so as not to exceed a "Southern U.S." incidental AEQ exploitation rate of 10.0% on Elwha chinook. Harvest at this level will assist recovery by providing adequate escapement returns to the river to perpetuate natural spawning in the limited habitat available, and provide broodstock for the supplementation program. It represents a significant decline in harvest pressure from southern U.S. fisheries. The SUS exploitation rate on the Strait of Juan de Fuca management unit aggregate averaged 33% for return years 1990 – 1996. Actual SUS AEQ exploitation rates for more recent years have not been calculated, however they were projected to be 7%, 5.0%, 5.2%, 4.8% and 4.7% respectively, in the final pre-season FRAM simulation models for management years 1999 through 2003.

The low abundance threshold for the Elwha River is 1,000 spawners, which represents a composite of 500 natural and 500 hatchery spawners. Whenever spawning escapement for this stock is projected to be below these levels, SUS fisheries will be managed to further reduce incidental AEQ mortality to less than 6.0%.

Data Gaps

- Estimates of total and natural smolt production from the Elwha River.
- Estimates of the age composition and description of life history of smolts.

Status Profile for the Western Strait of Juan de Fuca Management Unit

Component Stocks

Hoko River fall chinook

Geographic description

Fall chinook spawn primarily in the mainstem of the Hoko River, from above intertidal zone to RM 22, but primarily between RM 3.5 (the confluence of the Little Hoko River) to the falls at RM 10. Chinook may ascend the falls and spawn in the upper mainstem up to RM 22, and the lower reaches of larger tributaries such as Bear Creek (RM 0 to 1.2) and Cub Creek (RM 0 – 0.8), Ellis Creek (0 – 1.0), the mainstem (RM 0 – 2.5) and North Fork (RM 0 – 0.37), of Herman Creek, and Brown Creek(0 – 0.8). Chinook also spawn in the lower 2.9 miles of the Little Hoko River. Historically, chinook have also spawned in other Western Strait streams, including the Pysht, Clallam, and Sekiu rivers. Recent surveys of the Sekiu counted 52 and 12 chinook in 1998 and 1999, respectively. Their origin is unknown, but they are assumed to be strays from the Hoko system.

Currently, chinook from the Hoko Hatchery are being outplanted into the upper Hoko mainstem and tributaries of the upper and lower portions of the watershed, to seed high quality habitat, which has not been utilized consistently for spawning or rearing. Re-introduction to the Sekiu River, and other western Strait streams that once supported chinook, is also being planned.

Life History Traits

Based on scales collected from natural spawners and broodstock from 1988 – 1999, returning Hoko River adults are predominately age 5 (49%) and age 4 (31%) , with age 3 and age 6 adults comprising 8% and 10%, respectively, of the mean annual return (MFM 2000. The available data suggest that most smolts produced in the Hoko system emigrate as subyearlings (Williams et al. cited in Myer et al. 1998).

Status

The established escapement goal for Hoko River chinook is 850 natural spawners. This goal, first presented in 1978 in WDF *Technical Report 29*, is based on early estimates of freshwater habitat capacity. The total escapement goal is 1,050, which includes 200 brood stock for the supplementation and reintroduction program. For the Hoko chinook stock as a whole, the combined spawning escapement (natural plus hatchery) has averaged 1,243 spawners in the past five years. Total returns to the river (terminal run size shown above) have exceeded 850 chinook in 8 of the last 15 years).

Numbers of natural chinook spawners have significantly increased since the inception of the supplementation program in 1982, from counts of less than 200, before hatchery supplementation was initiated, to exceeding the natural escapement goal of 850 in three out of the last six years (the 1997 to 2002 average is 1,052 natural spawners). While natural-origin recruits and the recent and overall escapements have shown increasing trends in abundance since the early 1980s, the proportion of natural-origin spawners relative to the proportion of hatchery-origin spawners has declined in recent years. Nearly half the Hoko River natural spawners in most years may be attributed to the supplementation program (MFM 2000). Despite the recent escapements that have exceeded the goal of 850 natural spawners,, this goal has only been achieved in four of the last 15 years (1988 to 2002; Table 1).

Table 1. Natural spawning escapement of chinook and hatchery broodstock removals from the Hoko River, 1988 – 2002.

Return Year	Natural Spawners	Hatchery Brood Stock	Total Escapement
1988	686	90	776
1989	775	67	842
1990	378	115	493
1991	894	112	1,006
1992	642	98	740
1993	775	119	894
1994	332	96	428
1995	750	155	905
1996	1,228	37	1,265
1997	765	126	891
1998	1,618	104	1,722
1999	1,497	191	1,688
2000	612	119	731
2001	768	178	946
2002	443	237	680
1997 – 02 Avg	1,052	191	1,243
Goal:	850	200	1,050

Although the escapement goals set in Technical Report 29 have been commonly accepted over the past two decades, it is not certain that the spawner level of 850 is the optimum chinook escapement level for the Hoko River. Further analysis of habitat suitability and usage should be conducted to determine whether spawning or rearing habitat limits chinook production in the Hoko. Additional years of cohort reconstruction may also shed light on the stock-recruitment relationship for Hoko chinook, which may lead to revision in the escapement goal.

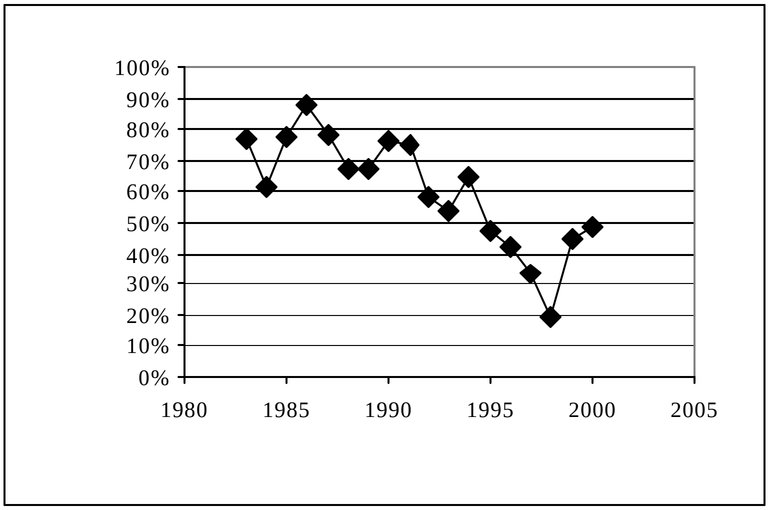
Harvest Distribution and Exploitation Rate Trends

The migration pathway, and harvest distribution, of Hoko River chinook has been described from recoveries of coded-wire tagged fish released from the Hoko Hatchery. The tag data suggest that Hoko chinook are harvested primarily by coastal fisheries in Southeast Alaska and British Columbia (Table 2).

Table 2. Harvest distribution of Hoko River chinook expressed as a proportion of total, annual, adult equivalent exploitation (CTC2003)

Years	Alaska	B.C.	Wash. Troll	Puget Sound Net	Washington sport
1997 - 2001	70.8%	26.5%	1.3%	0.1%	1.2%

Figure 1. Trend in total, adult equivalent, fisheries mortality for Juan de Fuca River chinook management units, estimated by post-season FRAM runs.



Post-season FRAM estimates indicate that the average annual exploitation rates for Juan de Fuca chinook units has declined 51 percent, from 1983-1987 to 1996-2000. These data are believed to correctly represent the trend for the Hoko River unit.

Although Hoko chinook were harvested at rates that should be reasonable for most Puget Sound chinook, even this exploitation rate was higher than would allow for replacement of spawners. This low productivity of Hoko chinook is very likely related to degraded freshwater habitat, including recurrent flooding and erosion, with poor marine survival. Almost the entire watershed (98%) has been clearcut, and 60% of the watershed is currently in a clearcut state (i.e., clearcuts <20 years old). There are 350 miles of roads in the 72 square mile watershed (M.Haggerty, Makah Fisheries Management, personal communication, 2000.)

Management Objectives

Management guidelines include a recovery exploitation rate objective for the Western Strait of Juan de Fuca management unit and a critical escapement threshold. The recovery exploitation rate objective is a maximum of ten percent in southern U.S. fisheries. It represents a lower exploitation rate than these stocks have experienced on average, and a rate that is achievable (and has been achieved in recent years), through conservative fishery management (Table 2). Recent years have shown that the nominal escapement goal can be achieved, with favorable marine survival, under this management regime.

The critical escapement threshold for the Hoko River is 500 natural spawners. Whenever natural spawning escapement for this stock is projected to be below this level, the harvest management plan will call for fisheries to be managed to achieve a lower rate than the interim 10% ceiling SUS exploitation rate.

Data gaps

- Reconstruct abundance of more recent brood years from CWT data
- Derive a spawner/recruit relationship for Hoko chinook

Appendix B. Non-landed Mortality

The fishery simulation model (FRAM) used by the co-managers for pre-season management planning and post-season assessment allows specification of non-landed mortality rates for different fisheries strata and gear types, in order to estimate total fisheries-related mortality for all component stocks. Non-landed mortality comprises a significant proportion of total fisheries mortality. This document summarizes the non-landed mortality rates that are currently specified by the FRAM chinook model (Table 1), and discusses the sources of these rates

When sub-legal fish (i.e. those less than the minimum allowable size) or species for which retention is disallowed are caught, a proportion (i.e. the releases mortality rate) subsequently die. This occurs frequently in commercial troll and recreational hook-and-line fisheries, for which regulations specify a minimum size limit, and may specify, for certain period, non-retention of chinook or coho. Non-retention of chinook may also be specified for certain net fisheries, where the fisherman tends the gear constantly (gillnets), or the gear design (seines) allows live capture and release of non-target species.

Drop-off or drop-out mortality is defined as that which occurs when fish are hooked or entangled by the gear, but they escape before being landed. The rate is applied to the number of landed fish.

Table 1 - Chinook Incidental Mortality Rates Assumed for FRAM Model Fisheries in Washington.

Fishery	Release Mortality	Drop-off, Drop-out, and other
Ocean Recreational	14%	5%
Ocean Troll – barbless hooks Barbed hooks	26% 30%	5% 5%
Puget Sound Recreational	> 22” 10% < 22” 20%	5% 5%
Gillnet Skagit Bay	 52.4%	2% terminal; 3% preterminal
Purse Seine	45% immature 33% mature	0%
Beach Seine Skagit Bay pink fishery	 50%	
Reef Net	None Assessed	0%

Ocean troll and recreational fisheries

Sources of Incidental Mortality

Incidental mortalities in troll fisheries are related to the duration of retention and non-retention periods, size limit regulations, and gear type. Size limits have been used extensively for these fisheries and have changed only a few times since 1979. Recreational and troll fisheries have been allowed to retain fish larger than 24” since the mid- 1980s. Troll fishing techniques differ, depending on whether the target species is chinook of coho. When coho are targeted, encounters with chinook have been reduced, but not eliminated, by species-specific gear, location, and fishing technique. Other management measures to reduce incidental chinook catch, such as landing limits, ratio fisheries, or chinook non-retention fisheries are seldom utilized. Marine mammal predation, ‘sorting’, and other sources of mortality associated with hook and line gear are not accounted in FRAM. ‘Sorting’ refers to release of legal fish in order to retain a larger fish later.

Estimates of Incidental Mortality

The effects of size limits on incidental mortality are modeled by a growth function to estimate what proportion of stock are of legal size at each time step. Encounter rates are calculated by the FRAM, using growth functions specific to each contributing stock to determine the proportion of legal and sub-legal fish, in each age class, present in each time step. Assuming that all ages are equally vulnerable to fishing, the fishery-specific exploitation rate is then applied to estimate legal and sub-legal encounters. Incidental mortality is then estimated by applying mortality rate appropriate to the fishery and gear type. FRAM also allows direct input of encounter rates if they are estimated from direct sampling of fisheries. With funding from the CTC, the Makah Tribe has monitored chinook encounter rates in troll fisheries in Washington Catch Areas 1 – 4 for 1998 - 2001. These data have been incorporated into pre-season fisheries modeling.

Release mortality associated with non-retention periods are calculated as ratios of non-retention days to normal retention days within the model base period. Drop-off mortality for hook-and-line fisheries is distinguished from landed catch by FRAM (i.e. may be reported separately). The current drop-off mortality rate is five percent. This value was derived from a negotiation process and is generally thought to include marine mammal interactions and illegal catch.

Historical estimates of incidental chinook mortality in troll and recreational fisheries, that are provided in the attached spreadsheets, were made by FRAM in ‘validation’ runs that reconstructed fisheries mortality, post-season, from known catch and stock abundance for the years 1983 – 1996. They are annual estimates, including impacts during the October – April time step that precedes the May – September period when most fishing occurs. These estimates express incidental mortality in the same terms as landed catch; they are not adjusted for adult equivalence. They provide a historical perspective on incidental mortality during the 1983-1985 base period, and under the more constrained fishing regimes of 1991 – 1996.

Measures to Reduce Incidental Mortalities

Incidental mortality has been reduced by requiring the use of barbless hooks in troll and recreational fisheries. During periods of chinook-directed fishing, trollers have been required to use large plugs to reduced interactions with sub-legal fish and coho. Time and area considerations are weighed in the structuring of ratio and non-retention fisheries to minimize incidental mortality to the extent possible.

Reduction of Incidental Mortality

Further reduction of incidental mortality in chinook fisheries will primarily be accomplished by measures designed to reduce encounters through time and area restrictions. The status of chinook stocks in Washington State may require reduction of exploitation rates. Future studies may show reductions in release mortality for different hook types and sizes for troll and recreational fisheries.

Net Fisheries

Sources of Incidental Mortality

Drift and set gillnet fisheries are conducted in Grays Harbor and Willapa Bay on the Washington coast, throughout Puget Sound, and in freshwater. However, net fisheries directed at chinook currently occur only in a few areas where harvestable, hatchery-origin chinook may be targeted. These areas include Bellingham Bay and the Nooksack River, Tulalip Bay, Elliot Bay and the Green River, the Puyallup River, Nisqually River, southern Hood Canal and the Skokomish River, and other discrete areas in southern Puget Sound. Incidental mortality occurs in these fisheries as a result of net drop-out and marine mammal predation. Gillnet fisheries retain all fish because the mortality of released fish is believed to be high. Harbor seals and sea lions cause significant incidental mortality in many pre-terminal and terminal gillnet fisheries in Puget Sound, but this source is not accounted in current fishery models or planning.

Purse seine fisheries are conducted in Georgia Strait / Rosario Strait, Southern Puget Sound, and Hood Canal, and are primarily directed at sockeye, pink, coho, and chum salmon. The only seine fishery directed at chinook occurs in Bellingham / Samish Bay.
Incidental mortality, in the context of this discussion, results from injury or stress during capture, or from handling the fish in order to release them. Mortality may be immediate or may occur after some delay from injury or disease.

Non-Indian reef net fisheries that target sockeye and, in some years, coho salmon are conducted in Puget Sound catch areas 7 and 7A. In recent years they have been required to release all chinook salmon, but no associated incidental mortality has been accounted in fishery planning. Reef net hauls catch relatively few fish, and the gear and handling cause relatively minor injuries (e.g. stress, scale loss), so incidental mortality is thought to be very low.

Marine mammal interactions incur significant incidental mortality in many Puget Sound gillnet fisheries, but they have not been generally quantified. A limited number of area-specific studies provide some quantification (PNPTC 1986; 1988?)

Estimates of Incidental Mortality

Drop-out mortality for gillnet fisheries are accounted by FRAM as 3% of landed pre-terminal gillnet catch and 2% of terminal landed gillnet catch. Many factors affect the drop out rate, including mesh dimension, net material and hanging design, sea state, and the frequency of picking. Drop-out rates were derived by technical consensus among state and tribal biologists, because of lack of data from direct sampling. Gillnets fished in the traditional manner are assumed to have a release mortality of a hundred percent. Incidental mortality due to marine mammal predation is highly variable, but is thought to be substantial in many areas in Puget Sound. There has been no systematic sampling of these fisheries that might enable accurate quantification, though anecdotal evidence abounds, and there have been several efforts to document the incidence of scars on spawning chinook.

When chinook are released following capture in purse seine fisheries, immediate and delayed mortality is significantly lower for large chinook than for smaller chinook (Ruggerone and June 1996). Incidental mortality is accounted in the FRAM model as 45% for immature fish (i.e. those caught in fall coho and chum fisheries), and 33% for mature fish caught in sockeye and pink fisheries. Pre-season projections of encounters for any given fishery are based on historic catch, and differential mortality calculated for large and small fish and reported as part of landed mortality. Since FRAM aggregates the incidental mortality associated with all types of net gear for a given fishery, the expected distribution of catch among different gear types underlies the estimate. 'Drop-out' mortality is not accounted for purse seine, roundhaul seine, or beach seine fisheries.

Estimates of mortality in net fisheries, that were included in the previous transmittal to the CTC, were based on a study conducted by WDFW in 1976-1985 (Shepard 1987). Observed encounters per set were expanded to estimate mortality in chinook directed fisheries and encounters per landing in other fisheries. These estimates were previously reported to PSC, but vary widely from FRAM estimates due to differences in methodology. We suggest that FRAM estimates provide the most useful comparison between the base period and more recent year; these are provided in attached spreadsheets.

Estimates of gillnet drop-out mortality from the FRAM validation set, for 1979 – 1985, and 1991 - 1996, are reported for marine net fisheries in North and South Puget Sound, Strait of Juan de Fuca, Grays Harbor, and Willapa Bay. Mortality, during these intervals, in freshwater net fisheries is reported as 2% of the landed catch in each river. River fisheries in this report include the Nooksack, Skagit, Snohomish, Lake Washington (including the Ship Canal), Green, Nisqually, and Skokomish rivers in Puget Sound, and the Sooes, Quileute, Queets, and Quinault rivers on the Washington coast.

Release mortality from purse seine fisheries is hard to tease out of FRAM validation runs. It is calculated by spreadsheet outside of FRAM and input as part of the landed catch. For a given FRAM net fishery, release mortality is dependent on the relative volume of purse seine, beach seine, and gillnet catch; no additional release mortality is assigned to beach seine and gillnet catch.

Measures to Reduce Incidental Mortality

Incidental chinook mortality has been reduced in gillnet fisheries by time and area restrictions that restrict effort during the chinook migration period, which has been specifically defined for all Puget Sound fishing areas. When migration periods for other salmon species overlap, (e.g. for pink or coho salmon), fisheries directed at those species are shortened to reduce chinook encounters.

Commercial net fishers may reduce marine mammal interactions by using 'seal bombs' or may obtain permits to shoot harbor seals and sea lions in some cases.

Since 1973, non-Indian fishery regulations have required that purse seines incorporate a strip of larger mesh at the top of the bunt to allow immature chinook to escape. In 1996, the minimum gill net mesh size for chum fisheries was increased to 6-1/4 from 5-3/4 inch mesh, in order to reduce the incidental catch of immature chinook. In 1997 all purse seine fisheries required release of all chinook. Gillnet fisheries were allowed to retain chinook because release mortality is assumed to be 100%. In 1998 shoreline closures in Rosario Strait (Area 7) were adopted, designed to reduce impacts on chinook salmon while still providing opportunities during sockeye and pink-directed fisheries. In 1999 purse seines were required to use brailers or hand dip nets to remove salmon from seine nets during sockeye and pink salmon fisheries in 7/7A to reduce by-catch mortality (R. Bernard, WDFW, pers comm. October 19, 2000).

Future Reduction of Incidental Mortality

Further reduction in the incidental mortality of chinook in net fisheries will involve coordinated study and development of more selective gear, more effective release techniques, mitigation of marine mammal interactions, and, perhaps, reductions in fishing opportunity.

A study, funded under NMFS' Saltonstall-Kennedy program, is currently being conducted by WDFW to evaluate tangle nets as an alternative to conventional gillnet gear. Tangle nets are constructed of smaller-mesh, loosely hung, monofilament that catches salmon by the teeth or jaw, rather than behind the opercle and gills. Previous studies in British Columbia suggested that non-target species could be released from this gear with low associated mortality. Fishing power with respect to target species, and survival of non-target salmon species caught and released from tangle nets, are being analyzed at two sites in Puget Sound. It may be possible to improve the survival of chinook caught in purse seines with careful handling or by allowing fish to recover in a tank prior to their release.
In certain circumstances fishing opportunity, where species other than chinook are the target, may be further constrained, or planned to achieve a specific level of incidental mortality. These measures require accurate in-season monitoring to assess when the threshold of landed chinook catch has been achieved.

Appendix C. Minimum Fisheries Regime

Non-Treaty Ocean Troll and Recreational Fisheries:.

- Chinook and coho quotas and seasons adopted by the PFMC.
- Exploitation rates on critical Puget Sound Chinook management units will not exceed the range projected to occur for management years 2000 – 2003 (see Chapter 5).

Treaty Ocean Troll Fishery:

- Chinook and coho quotas and seasons adopted by the PFMC.
- Exploitation rates on critical Puget Sound Chinook management units will not exceed the range projected to occur for management years 2000 – 2003 (see Chapter 5).

Strait of Juan De Fuca Treaty Troll Fisheries:

- Open June 15 through April 15.
- Use barbless hooks only.

Strait of Juan De Fuca Treaty Net Fisheries:

- Setnet fishery for Chinook open June 16 to August 15. 1000-foot closures around river mouths.
- Gillnet fisheries for sockeye, pink, and chum managed according to PST Annex.
- Gillnet fisheries for coho from the end of the Fraser Panel management period, to the start of fall chum fisheries (approximately Oct. 10).
- Closed mid-November through mid-June.

Strait of Juan De Fuca Non-treaty Net Fisheries:

- Closed year-around.

Area 5/6 Recreational Fishery:

- May 1-June 30 closed.
- July 1 – Sept 30 Chinook mark selective fishery not to exceed two months, and not to exceed 3500 landed catch in 2004. In subsequent years, this may be extended by agreement of the co-managers, else, Chinook non-retention.
- October closed
- 1-Chinook bag limit in November.
- December 1 - February 15 closed
- 1-fish bag limit February 16-April 10
- April 11-30 closed

Strait of Juan De Fuca Terminal Treaty Net Fisheries:

- Hoko, Pysht, and Freshwater Bays closed May 1 – October 15.
- Elwha River closed April 1 through mid-September, except for minimal ceremonial harvests.
- Dungeness Bay (6D) closed March 1 through mid-September; Chinook non-retention mid-September – October 10.
- Dungeness River closed March 1 through September 30. Chinook non retention when open, except for minimal ceremonial harvests.
- Miscellaneous JDF streams closed March 1 through November 30.

Strait of Juan De Fuca River Recreational Fishery:

- June 1 – Sept 30 Elwha River closed to all fishing from river mouth to WDFW channel. At all other times and places, Chinook non-retention.
- Dungeness closed to salmon 12/1 through 10/15.
- Dungeness Chinook non-retention 10/16 through 11/30.
- Close other streams.

Area 6/7/7A Treaty and Non-treaty Net Fisheries:

- Sockeye, pink, and chum fisheries managed according to PST Annex.
- Net fisheries closed from mid-November through mid-June.
- Area 6A Closed.
- Non-treaty purse seine and reef net fisheries Chinook non-retention.
- Non-treaty gillnet fishery Chinook ceiling of 700.
- Non-treaty closure within 1500 feet of Fidalgo Island between Deception Pass and Shannon Pt; and within 1500 feet of Lopez and Decatur Islands between Pt Colville and James Island.

Area 7 Recreational Fishery:

- May 1-June 30 closed.
- 7/1-7/31 1 fish limit, Rosario Strait and Eastern Strait of Juan de Fuca
- closed; Bellingham Bay closed.
- 8/1-9/30 1 fish limit, Southern Rosario Strait and Eastern Strait Juan de
- Fuca closed Bellingham Bay closed.
- 8/1-8/15, Samish Bay closed.
- Chinook non-retention 10/1-10/31
- 11/1-11/30 1 fish limit.
- December-February 15 closed
- 1-fish bag limit February 16-April 10
- April 11-30 closed

Nooksack/Samish Terminal Area Fisheries:

- Bellingham Bay (7B) and Samish Bay (7C) closed to commercial fishing from April 15 through July 31.
- Area 7B/7C hatchery fall Chinook fishery opens August 1.
- Pink fishery opens August 1.
- Ceremonial fishery in late May limited to 10 natural-origin Chinook.
- Subsistence fishery limited 20 natural-origin Chinook between July 1-4.
- Ceremonial and subsistence harvest to be taken in the lower river, and between the confluence of the South Fork and the confluence of the Middle Fork.
- Nooksack River commercial fishery for hatchery fall Chinook opens August 1 in the lower river section; and staggered openings in up-river sections will occur over 4 successive weekly periods. (see Appendix A).
- Bellingham Bay recreational fishery closed in July.
- Samish Bay recreational fishery closed August 1-15.
- Chinook non-retention in Nooksack River recreational fisheries.
- 2-Chinook bag limit after October 1 in Nooksack River.
- 2-fish bag limit from July 1 to December 31 in Samish River.

Skagit Terminal Area Net Fisheries:

- Skagit Bay and lower Skagit River closed to commercial net fishing from mid-February to August 22 in pink years, and until week 37 (~September 10) in non-pink years.
- Upper Skagit River closed to commercial net fishing from mid-March to August 22 in pink years, and until week 42 (~October 10) in non-pink years, unless there is an opening for Baker sockeye in July.
- Upper Skagit and Sauk-Suiattle fisheries on Baker sockeye require 5½ "
- maximum mesh, and Chinook non-retention.
- Half of the Upper Skagit and Sauk-Suiattle share of Baker sockeye will be taken at the Baker Trap, rather than in river fisheries.
- No Chinook update fishery or directed commercial Chinook fishery.
- Treaty pink update fishery limited to 2 days/week during weeks 35 and 36, and Non-treaty update limited to 1 day/week, gillnets only.
- Pink fishery gillnet openings in the Skagit River limited to a maximum of 3 days/week, regardless of pink numbers. Beach seines may be used on other days, with Chinook non-retention.
- Up to 40% of the Upper Skagit share of pink salmon will be taken in Skagit Bay.
- Release Chinook from beach seines in Skagit Bay.
- Chinook non-retention required in pink fisheries in the upper river.
- Tribal coho openings delayed until Week 39 in the Bay and lower river, and until Week 42 in the upper river.
- Chinook test fisheries limited to 1 boat, 6 hrs/week.

Skagit River Recreational Fisheries:

- Chinook non-retention.

Area 8A and 8D Net Fisheries:

- Area 8A Treaty fishery Chinook impacts incidental to fisheries directed at coho, pink, chum, and steelhead.
- Effort in the Treaty pink fishery will be adjusted in-season to maintain Chinook impacts at or below those modeled during the pink management period.
- Area 8D Treaty Chinook fisheries limited to C & S beginning in May, and to 3 days/wk during the Chinook management period.
- Non-treaty pink fishery limited to 1 day/week for each gear.
- Non-treaty purse seine fishery Chinook non-retention.
- Area 8D non-treaty Chinook impacts incidental to fisheries directed at coho and chum.

Stillaguamish River Net Fisheries:

- Treaty net fishery Chinook impacts incidental to fisheries directed at pink, chum, and steelhead.
- Treaty pink fishery schedule limited to maintain Chinook impacts at or below the modeled rate.

Stillaguamish River Recreational Fisheries:

- Chinook non-retention.
- Use barbless hooks from September 1 to December 31.

Snohomish River Fisheries:
- Net fisheries closed.
- Chinook non-retention in river recreational fisheries.

Area 8-1 Recreational Fisheries:
- 5/1-8/31 closed.
- Chinook non-retention 9/1-10/31.
- 11/1-11/30 1 fish limit.
- 12/1-2/15 closed.
- 1-fish bag limit February 16 – April 10.
- 4/11-4/30 closed.

Area 8-2 Recreational Fisheries:
- 5/1-7/31 closed.
- Chinook non-retention 8/1-10/31.
- 11/1-11/30 1 fish limit.
- 12/1-2/15 closed.
- 1-fish bag limit February 16 – April 10.
- 4/11-4/30 closed.
- 1-Chinook bag limit in Tulalip Bay in August and September.
- Tulalip Bay openings limited to 12:01 AM Friday to 11:59 AM Monday each week.

Area 9 Net Fisheries:
- Net fisheries limited to research purposes.

Area 9 Recreational Fisheries:
- 5/1-7/31 closed.
- Chinook non-retention 8/1-10/31.
- 11/1-11/30 1 fish limit.
- 12/1-2/15 closed.
- 1-fish bag limit February 16 – April 10.
- 4/11-4/30 closed.

Area 10 Net Fisheries:
- Closed from mid-November through June and August.
- Sockeye net fishery during first three weeks of July when ISU indicates harvestable surplus of Lake Washington stock.
- Net fisheries for coho and chum salmon will be determined based on in-season abundance estimates of those species. Limited test fisheries will begin the 2nd week of September. Commercial fisheries schedules will be based on effort and abundance estimates. Marine waters east of line from West Point to Meadow Point shall remain closed during the month of September for Chinook protection. Chinook live release regulations will be in effect

Lake Washington Terminal Area Fisheries:

- Chinook run size update from lock count to re-evaluate forecasted status.
- No Chinook directed commercial fishery in the Ship Canal or Lake Washington.
- Net fishery impacts incidental to fisheries directed at sockeye and coho. Sockeye and coho fisheries dependant on lock count ISU. Incidental Chinook impact minimized by time, area and live Chinook-release restrictions. Sockeye fisheries scheduled as early as possible. Coho fishery delayed until September 15th when 95.2% of the Chinook run has cleared the locks.
- Possible directed Chinook fishery in Lake Sammamish for Issaquah Hatchery surplus.
- Cedar River and Issaquah Creek closed to recreational fishing.
- Chinook non-retention in Sammamish River, Lake Washington, Lake Union, Portage Bay, and Ship Canal recreational fisheries

Area 10A Treaty Net Fisheries:

- Chinook gillnet test fishery 12 hours/week, 3 weeks, beginning mid-July to re-evaluate forecasted status.
- No Chinook directed commercial fishery.
- Net fishery impacts incidental to fisheries directed at coho. Coho opening delayed until September 15th.

Duwamish/Green River Fisheries:

- Commercial Chinook fishery dependant on Area 10A test fishery results.
- No Chinook directed commercial fishery.
- Net fishery impacts incidental to fisheries directed at coho. Coho opening delayed until September 15th and restricted to waters below the 16th Ave Bridge. Coho opening above the 16th Ave Bridge to the turning basin delayed until September 22nd. Coho opening above the turning basin up to the Hwy 99 Bridge delayed until September 29th.
- Chinook non-retention in river recreational fisheries

Area 10E Treaty Net Fisheries:

- Closed from mid November until last week of July.
- Chinook net fishery 5 day/wk last week of July through September 15.
- Chinook impacts incidental to net fisheries directed at coho and chum, from mid-September through November

.Area 10 Recreational Fisheries:

- 5/1-6/30 closed.
- Chinook non-retention 7/1-10/31.
- 11/1-11/30 1 fish limit.
- 12/1-2/15 closed.
- 1-fish bag limit February 16 – April 10.
- 4/11-4/30 closed.

Area 11 Net Fisheries:

- Closed from end of November to beginning of September.
- No Chinook-directed fishery
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.
- Non-treaty purse seine fishery Chinook non-retention.

Area 11A Net Fisheries:
- Closed from beginning of November to end of August.
- Net fishery Chinook impacts incidental to fisheries directed at coho.

Puyallup River System Fisheries:
- Net fisheries closed from beginning of February to beginning of August.
- Limit gill net test fishery for Chinook to 1 day a week, scheduled from mid-July through August 15.
- Chinook net fisheries limited to 1 day/week, August 15 – September 10 (delayed to protect White River spring Chinook.
- Muckleshoot on-reservation fisheries on White River limited to hook and line C & S fishing for seniors, with a limit of 25 Chinook.
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.
- 2-Chinook bag limit in river sport fisheries.
- Chinook non-retention before August 1 in Puyallup River sport fishery.
- Chinook non-retention before September 1 in Carbon River sport fishery.
- Chinook non-retention in White River.

Area 11 Recreational Fisheries:
- 5/1-5/30 closed.
- 1-fish limit June 1 – November 30.
- 12/1-2/15 closed.
- 1-fish limit February 16 – April 10.
- 4/11-4/30 closed.

Fox Island/Ketron Island Net Fisheries:
- Closed from end of October to August 1.
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.

Sequalitchew Net Fisheries:
- Net fishery Chinook impacts incidental to fisheries directed at coho.

Carr Inlet Net Fisheries:
- Closed from beginning of October through August 1.
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.

Chambers Bay Net Fisheries:
- Closed from end of mid-October to August 1.
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.

Area 13D Net Fisheries:
- Closed from mid-September to August 1.
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.

Henderson Inlet (Area 13E) Net Fisheries:
- Closed year-around.

Budd Inlet Net Fisheries:
- Closed from mid-September to July 15.
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.

Areas 13G-K Net Fisheries:
- Closed Mid-September to August 1.
- Net fishery Chinook impacts incidental to fisheries directed at coho and chum.

Nisqually River and McAllister Creek Fisheries:
- Chinook fishery late-July through September, up to three days per week dependent on in-season abundance assessment (see Appendix A).
- Coho fishery October through mid-November.
- Late chum fishery mid-December – mid-January.
- Nisqually River recreational closed February 1 through May 31.
- McAllister Creek recreational closed December 1 through May 31.
- Chinook non-retention in June recreational fishery.
- 2-Chinook bag limit.

Area 13 Recreational Fisheries:
- 1-fish bag limit May 1-November 30.
- 12/1-2/15 closed.
- 1-fish bag limit February 16 – April 10.
- 4/11-4/30 closed.

Hood Canal (12, 12B, 12C, 12D) Treaty Net Fisheries: (also see: Skokomish and Mid-Hood Canal Management Unit profiles in Appendix A):
- Chinook directed treaty fishery limited to Areas 12C and 12H.
- Coho directed fisheries in Areas 12 and 12B delayed to Sept. 24; in Area 12C, to Oct. 1. Beach seines release Chinook through Oct. 15.
- 1,000 foot closures around river mouths, when rivers are closed to fishing.
- Net fisheries closed from mid December to mid July

Area 9A Treaty Net Fisheries:
- Closed from end of January to mid-August (dependent upon pink fishery).
- Beach seines release Chinook through Oct. 15.

Area 12A Treaty Net Fisheries:
- Closed from mid-December to mid-August.
- During coho and chum fisheries, beach seines release Chinook through Oct. 15.

Hood Canal Freshwater Treaty Net Fisheries:
- Dosewallips, Duckabush, and Hamma Hamma rivers closed.
- Skokomish River Chinook fishery August 1 – September 30, limited to two to five days per week.
- Skokomish River closed March – July 31(also see: Skokomish MU profile in Appendix A).

Area 12 Recreational Fishery:

- 5/1-6/30 closed.
- Chinook non-retention 7/1-10/15.
- 10/16-12/31 1-fish limit.
- 1/1-2/15 closed.
- 1-fish bag limit February 16 – April 10.
- 4/11-4/30 closed.

Hood Canal Freshwater Recreational Fisheries:

- Closed March 1 to May 31.
- Chinook non-retention from June 1 to February 29 in all rivers.
- Dosewallips, Duckabush, and Hamma Hamma closed in September and October.

Appendix D. Role of Salmon in Nutrient Enrichment of Fluvial Systems

INTRODUCTION

Continued declines in abundance of Pacific salmon (*Oncorhynchus* spp.) populations have focused increased attention on factors limiting their survival. While the decline in abundance of Pacific salmon stocks (National Research Council 1996) has been attributed to may factors, just recently have researchers focused their attention on the nutrient re-cycling role of returning adult salmon in maintaining productive freshwater ecosystems. Given that Pacific salmon accumulate the significant majority of their body mass while in the marine environment (Groot and Margolis 1991), returning runs of adult salmon potentially represent a substantial source of marine-derived nutrients (MDN) for freshwater and riparian communities (Larkin and Slaney 1996; Gresh et al. 2000; Murota 2002; Schoonmaker et al. 2002). Research has shown that the addition of nutrients to freshwater systems can influence community structure and increase stream productivity at several trophic levels (Kline et al. 1990; Piorkowski 1995; Quamme and Slaney 2002). Benefits include increased growth and density of juvenile salmonid populations (Johnston et al. 1990; Bradford et al. 2000; Ward and Slaney 2002). Gresh et al. (2000) estimate that the current contribution of MDN from adult Pacific salmon to rivers in the Pacific Northwest is as low as 6-7% of historic levels and that the resulting 'nutrient deficit' could be exacerbating continued declines in salmon abundance or impeding recovery.

The concept of a 'nutrient deficit' has several implications for current fisheries management, harvest strategies and recovery of depressed salmon stocks. It is asserted that current harvest management strategies for salmon stocks fail to consider the importance of MDN for maintaining properly functioning ecosystems and self-sustaining salmon populations (Micheal 1998; Cederholm et al. 2000; Gresh et al. 2000; Bilby et al. 2001). More directly, current escapement goals for salmon runs may be perpetuating a negative feedback loop in salmon population dynamics (Larkin and Slaney 1996, 1997). Ideally, research might quantify the nutrient input, and escapement density, necessary to optimize ecosystem function, viable salmon runs, and harvest. However, nutrient dynamics in aquatic systems are often complex (Northcote 1988; Polis et al. 1997; Bisson and Bilby 1998; Murphy 1998; Naiman et al. 2000) and depend on numerous site-specific factors including the species of salmon, spawning density and location, stream discharge regimes, stream habitat complexity, basin geology, light, temperature and community structure. Researchers are just beginning to recognize and understand these complexities in relation to salmon and MDN. In this paper I will review the current state of knowledge on the relationship between Pacific salmon, MDN and stream ecosystem function in the context of determining 'ecologically based' salmon escapement goals.

NUTRIENT PATHWAYS

Adult salmon contain proteins, fats and other biochemicals comprised of marine- origin carbon, nitrogen and phosphorous (Mathisen et al. 1988). Returning adult salmon act as vectors in delivering nutrients of marine origin to terrestrial ecosystems through excretion (O'Keefe and Edwards 2002), gametes and carcasses (Mathisen et al. 1988). In general, stream biota incorporate salmon-derived nutrients through three primary pathways: 1) trophic transfer following uptake of inorganic nutrients by primary producers; 2) streambed microfaunal uptake of dissolved organic matter released by salmon carcasses; and 3) direct consumption of salmon carcasses, eggs and fry (Cederholm et al. 1999). Additionally, high flow events and scavenging by birds and mammals (Cederholm et al. 1989, 2000; Ben-David et al. 1998) can deliver salmon-derived nutrients to riparian and upland communities (Garten 1993; Wilson and Halupka 1995; Helfield and Naiman 2001; Hocking and Reimchen 2002; Reimchen et al. 2002).

STABLE ISOTOPE AND PROTEIN STUDIES

Applied relatively recently to the issue of salmon and MDN, stable isotope analysis has allowed researchers to quantitatively identify nutrient sources and further understand nutrient pathways in freshwater systems. Carbon, nitrogen, and phosphorous are typically considered principal nutrients that limit ecosystem productivity (Gregory et al. 1987; Peterson and Fry 1987; Murphy 1998). While phosphorous has only one stable isotope, limiting our ability to distinguish the origin of phosphorous, carbon (C) and nitrogen (N) have two stable isotopes. The isotopic properties of carbon and nitrogen provide natural tracers for determining differences in stable isotope abundance in trophic food webs. Stable isotope ratios are typically expressed as $\delta^{13}C$ and $\delta^{15}N$ values and represent the level of enrichment or depletion of the heavier isotope C or N relative to a standard (Peterson and Fry 1987). Spawning salmon contain higher proportions of the heavy isotopes carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$, Mathisen et al. 1988; Piorkowski 1995; Bilby et al. 1998). Nitrogen is especially applicable in salmon-derived nutrient studies due to the dichotomous nature in N sources between Pacific salmon (oceanic N) and terrestrial and freshwater systems (atmospheric N_2, Peterson and Fry 1987; Kline et al. 1997).

Kline et al. (1990) developed an isotope-mixing model to investigate the incorporation of MDN in Sashin Creek, southeastern Alaska. The isotope-mixing model allows for determination of percent contribution of marine nitrogen across trophic levels. The study design compared isotope ratios between a lower reach, accessed primarily by pink salmon (approximately 30,000 adults annually), and an upper control reach isolated from anadromous fish. Isotope values indicate that standing crop of periphyton in the anadromous section was dependent on marine N, with levels greater than 90% immediately after spawning and near 50% at other times of the year. The sustained marine N signal in periphyton further indicated nutrient retention. Stonefly nymphs and caddis fly larvae also showed high levels of enrichment in April possibly due to overwintering retention and trophic transfer through periphyton and decomposers (e.g. fungi). The isotope model suggested that turbellarians were incorporating marine N through direct consumption of salmon eggs. In rainbow trout, high levels of $\delta^{15}N$ were found with increasing isotope values as the size of trout increased. Using a dual isotope method, Kline et al. (1990) concluded that trout from the enriched section were likely incorporating a portion of marine N from autochthonous production (dependent on primary producer uptake of remineralized nutrients) as well as direct feeding on salmon carcasses and eggs. Researchers surmise that MDN have a trophic-wide effect in the anadromous section of Sashin Creek. They also note that the use of fertilizers to alleviate nutrient loss in streams may not adequately substitute for salmon carcasses and eggs that are directly fed upon by consumers and decomposers, a point further developed in this review.

Since the Kline et al. (1990) study, numerous investigators have used stable isotope methods to distinguish MDN pathways in lotic systems (Bilby et al. 1996, 1998, 2001; Helfield and Naiman 2001; Piorkowski 1995; Winter et al. 2000). These studies show similar results indicating incorporation of MDN in food webs with anadromous runs of salmon. However, results do not universally indicate the degree of importance or pathways of MDN across different lotic systems. In an in-depth ecosystem study on five creeks in southcentral Alaska, Piorkowski (1995) used stable isotopes to distinguish marine N in stream food webs. The five study creeks are used by multiple species of anadromous salmon of which Piorkowski (1995) found different isotopic composition between adult salmon species with chinook salmon being significantly more enriched in $\delta^{15}N$ (due to increased ocean residence time) as compared to pink, coho and chum salmon. Isotope samples were collected from organisms at several trophic levels. Samples from sites with adult salmon returns indicated that the diets of grayling, rainbow trout, and coho salmon fry were predominately comprised of salmon tissue and eggs. Also, examination of stream macroinvertebrates revealed increased taxa richness and diversity in anadromous stream sections compared with non-anadromous sections. Despite this, results failed to detect a significant marine N signal between control and treatment sites in samples of riparian vegetation, algae, and stream macroinvertebrates (grazers) and implies that marine N was not significantly incorporated through pathways of primary production. Piorkowksi (1995) notes that results markedly differ from the Sashin Creek study (Kline et al. 1990) and are likely due to two important considerations: 1) Sashin Creek received a much larger run of salmon utilizing a smaller stream area; and 2) total dissolved nitrogen content in Sashin Creek was likely much lower given intense precipitation (nutrient flushing), causing the system to be more dependent on seasonal pulses of salmon-derived nutrients.

Many headwater streams in the Pacific Northwest exhibit low levels of primary and secondary productivity (Gregory et al. 1987; Bilby and Bisson 1992), and are systems typically preferred by adult coho salmon for spawning (Sandercock 1991). Bilby et al. (1996) compared isotope ratios in four tributaries of the Snoqualmie River, Washington, to determine the influence of coho salmon carcasses on food webs of headwater streams. Overall, the study suggests that even modest inputs of MDN can influence small streams. $\delta^{15}N$ and $\delta^{13}C$ values were similar between anadromous and non-anadromous streams prior to coho salmon spawning; during and shortly after spawning, elevated $\delta^{15}N$ values were found in stream biota (epilithic organic matter and stream invertebrates) and riparian foliage. Juvenile coho salmon more than doubled their weight following the appearance of spawning adults. Using an isotope model assuming no direct consumption on salmon carcasses and eggs (resulting in a conservative estimate without trophic fractionation), juvenile coho salmon were enriched approximately 30% with marine N. As well, researchers found rapid uptake of MDN through chemical sorption by streambed gravel. Chemical uptake of dissolved organic matter by streambed substrate was similar in both light and dark controlled experiments. Bilby et al. (1996) stress the importance of chemical sorption for initial nutrient uptake in headwater streams where primary production is limited during winter due to cold temperatures, low light levels, and frequent scouring by high flow events.

Carcass tissue and eggs appear to be an important food source for juvenile fish during winter periods and may play a critical role when other food items are less available. In four streams in southwestern Washington, Bilby et al. (1998) observed significant increases in density, weight and condition factor of juvenile steelhead and coho salmon following addition of hatchery spawned coho carcasses (with some eggs remaining). In enriched stream sections, 60-96% of stomach contents of juvenile steelhead and coho salmon were comprised of carcass flesh and eggs (with eggs being the preferred food item) while carcass material was present. Also, diet content of juvenile coho salmon had five times the amount of invertebrate biomass as compared to non-enriched areas. While significant increases in density and condition factor of juvenile coho salmon and steelhead were observed in carcass enriched areas, fish were not marked to confirm site fidelity throughout the study period. Even so, increased fish size and condition factor has implications for higher survival for both juvenile coho salmon (Bell 2001; Brakensiek 2002; Hartman and Scrivener 1990; Quinn and Peterson 1996; Holtby 1988) and steelhead (Ward and Slaney 1988) and subsequent returns of adults (Hager and Noble 1976; Bilton et al. 1982).

Findings by Wipli et al. (in review) further corroborate conclusions by Bilby et al. (1998) on the importance of salmon carcasses and eggs for juvenile coho salmon. In experimental and natural streams in Southeast Alaska, Wipfli et al. (in review) found strong positive correlations between salmon carcass loading rates and growth of juvenile coho salmon, cutthroat trout and Dolly Varden char. Over a 60 day experiment, juvenile coho salmon gained over 60% of fish body mass in study reaches with the highest carcass loading rates (4 carcasses / m^2). Similarly, cutthroat trout and Dolly Varden char exhibited growth rates over five times higher in carcass rich areas as compared to control areas. Nutritional status of juvenile coho salmon was evidenced by concentrations of triacylglyceride (TAG) and ratios of marine-based to terrestrial-based fatty acids in juvenile samples; both percent TAG and fatty acid ratios increased with increasing density of carcasses. TAG concentrations in juvenile fish correspond to storage of marine-derived long-chain n-3 fatty acids and indicates direct benefits of salmon carcasses to growth and nutritional status of stream salmonids.

BOTTOM-UP EFFECTS OF NUTRIENT ENHANCEMENT

Studies reviewed thus far indicate that stream delivery of MDN and biogenic material from returning adult salmon provide an immediate food resource for fish and can influence lotic food webs. Addition of nutrients can certainly have a bottom-up effect in freshwater systems, boosting primary production and ultimately benefiting fish populations (Johnston et al. 1990; Bradford et al. 2000; Ward et al. 2002; Wilson et al. 2002). This management concept has seen successful application in lake enrichment programs in Alaska and British Columbia where returning runs of sockeye salmon have increased as a result of manual application of nutrients. The extensive knowledge and management success in sockeye rearing lakes is due, in part, to the relative simplicity of these systems in food web and nutrient dynamics, as compared to fluvial systems (Kline et al. 1997; Kyle et al. 1997). Sockeye salmon rearing lakes have generally been identified as oligotrophic systems, primarily limited by phosphorous. Ratio additions of nitrogen and phosphorous have successfully elevated lake rearing capacities for juvenile sockeye salmon through increased zooplankton production (Hyatt and Stockner 1985; Kyle et al. 1997; Bradford et al. 2000). British Columbia has carried this management tool further and begun fertilizing large river systems in efforts to boost declining steelhead and coho salmon populations. Results so far show overall stimulation of system productivity with increased density and growth of juvenile coho salmon and steelhead as well as earlier age at outmigration of steelhead (Johnston et al. 1990; McCubbing and Ward 2000; Ward and Slaney 2002). Whether manual fertilization of large river systems can recover coho salmon and steelhead runs remains to be seen. While certainly a management and research tool, it is questionable if manual nutrient supplementation programs can adequately replace ecosystem function of spawning adult salmon.

Examples of manual supplementation studies are raised to illustrate issues of trophic capacity in relation to fish production. Productivity can be defined as the capacity of a system to produce a product of interest (Bisson and Bilby 1998). A nutrient limited system can mean food limited in the interest of fish production (Chapman 1966; Dill et al. 1981; Johnston et al. 1990). While adult salmon carcasses and eggs provide a direct food resource for fish populations, salmon-derived nutrients can potentially influence fish production through autotrophic and heterotrophic pathways as well (see Vannote et al. 1980, Bilby and Bisson 1992). Wipfli et al. (1998) conducted highly replicated tests of adding salmon carcasses in experimental and natural stream channels in Alaska to assess responses in primary production. Biofilm production (a food source for aquatic invertebrates) increased approximately 15 times in the carcass enriched section (with an approximate return run size of 75,000 pink salmon) compared to the upstream control section. Further, total macroinvertebrate densities increased up to 8 and 25 times in artificial and anadromous stream sections, respectively, as compared to control sections. Similar results were found in a follow-up study by Wipfli et al. (1999), and also suggest a threshold level of response in biofilm production (over a two-month study period) in relation to carcass loading rates (up to 1.45 kg, the lowest carcass loading rate in artificial channels). Both studies (Wipfli et al. 1998, 1999) show trophic responses to MDN and suggest potential growth benefits to fish through increased availability of fish food organisms (see also Perrin et al. 1987, Johnston et al. 1990, Perrin and Richardson 1997, Quamme and Slaney 2002). Wipfli et al. (1999) caution however, that the capacity for stream systems to retain marine nutrients and the long-term effects of 'excessive' carcass loadings for stream productivity have yet to be sufficiently addressed by researchers (O'Keefe and Edwards 2002).

STREAM RETENTION OF SALMON CARCASSES

Stream incorporation of marine-derived nutrients necessitates that salmon carcasses are retained for a sufficient period of time. Cederholm and Peterson (1985) investigated winter retention of coho salmon carcasses in several small streams on the Olympic Peninsula in western Washington. They initially released 180 carcasses throughout nine streams with varying abundance of large woody debris. One week following releases, 78 (43%) of the study carcasses were identified of which 80% were within 200 m of initial placement. Carcass retention was positively correlated with increases in large woody debris. The researchers speculated that carcass retention could be even higher in unlogged streams where large woody debris loading was higher as compared to their study streams.

In a similar follow-up study on carcass retention in Olympic Peninsula streams, Cederholm et al. (1989) released 945 tagged coho salmon carcasses, of which 174 were implanted with radio transmitters to more definitively determine the fate of mobilized carcasses. Few study carcasses were flushed beyond 600 m with a median travel distance of 49.5 m from initial placement. Again, large woody debris was influential in retaining salmon carcasses with the majority of carcasses found in pools. Cederholm et al. (1989) also assessed retention during high flows by depositing 25 radio-tagged carcasses at the beginning of a flood event (estimated discharge 6.20 m^3/s). Following the flood event, 21 of the 25 radio-tagged fish were located within 600 m of initial placement, with a median travel distance of 66 m. Ten of the radio-tagged carcasses were found on stream banks well above low flow levels. In a different study, Glock et al. (1980) investigated retention of chum salmon carcasses on a much larger system, the Skagit River in Washington. Although carcasses drifted as far as 39 km within the first five days, the majority of carcasses (20%) were located within 1.5 km of initial placement. Habitat, discharge, amount of large-woody debris, and species of salmon appear to be important factors in considering retention of salmon carcasses in fluvial systems.

The study by Cederholm et al. (1989) also revealed significant predation by mammals and birds on salmon carcasses. Approximately 22 taxa of mammals and birds were documented consumers of salmon carcasses. Surveys identified 374 partially eaten study carcasses removed from stream channels with 88% of these carcasses located within 15 m of the stream bank. Cederholm et al. (2000) provide a more extensive review of wildlife-salmon relationships that documents over 138 species having a 'strong' positive life-history relationship to Pacific salmon. This and other research suggests the ecological relationships between salmon and wildlife (Wilson and Halupka 1995; Ben-David et al. 1998; Wilson et al. 1998). Further, wildlife species appear to play a significant role in the removal of salmon carcasses from lotic systems where nutrient benefits may be more realized in riparian and upland communities (Cederholm et al. 2000; Garten 1993; Helfield and Naiman 2001; Reimchen et al. 2002).

IMPLICATIONS FOR FISHERIES MANAGEMENT

Although research to date provides evidence of the role of salmon-derived nutrients in ecosystem function, this complex relationship is poorly understood. Further understanding of the ecosystem context of returning adult salmon and MDN will require both the synthesis of several scientific disciplines and human values. Given the high cultural and economic value of salmon, and the public mandate to recover natural salmon populations, fisheries managers must insure that harvest practices do not impede recovery. Research on salmon and MDN frequently implies that current harvest management strategies exacerbate the risk of further decline in salmon populations, due to removal of salmon and nutrients bound for terrestrial systems. However, the science of quantifying salmon escapement goals necessary to properly functioning ecosystems is still in infancy.

Nonetheless, research is beginning to focus on quantifying nutrient input levels necessary to improve juvenile salmon survival. Bilby et al. (2001) used stable isotope levels from juvenile coho salmon collected throughout western Washington to test for a marine N threshold level in juvenile fish. Representative of 26 stream reaches from 12 different watersheds, juvenile coho salmon samples were collected in late February and early March over a seven-year period. Juvenile samples were only collected in known areas where no other anadromous fish spawn. Cutthroat trout were collected above anadromous barriers in the same systems that juvenile coho salmon samples were collected. Isotope values from cutthroat trout represented $\delta^{15}N$ background levels used to establish site-specific ratio index measures of marine N enrichment in relation to $\delta^{15}N$ values from juvenile coho salmon. Also, tissue samples were collected from hatchery returns of adult coho salmon throughout the region to relate $\delta^{15}N$ values from cutthroat trout and juvenile coho. Adult returns of coho salmon to each creek were determined using spawner count and stream habitat data; average weights from adult hatchery returns were used to estimate biomass (wet-weight kg / m^2) of spawners in each study creek.

Bilby et al. (2001) found that $\delta^{15}N$ values were consistently higher, by study site, for juvenile coho salmon as compared to cutthroat trout. However, isotope values revealed considerable variation between study streams for both cutthroat trout (ranging from 4.5‰ to 8.5‰, the per mil deviation of $^{15}N/^{14}N$ from air N_2, Peterson and Fry 1987; Kline et al. 1990) and juvenile coho salmon (5.8‰ to 11.7‰). Cutthroat $\delta^{15}N$ values suggest other sources of marine N, or possibly nutrient fractionation (Peterson and Fry 1987; Kline et al. 1990). Variation in isotope values reveals the need to establish basin-specific background isotope levels when using isotope methods.

Using the relationship between estimated carcass abundance and ^{15}N index values of enrichment in juvenile coho salmon, Bilby et al. (2001) found that enrichment levels increased with increasing carcass abundance. The relationship also revealed a point of diminishing enrichment of marine N in juvenile coho salmon above carcass abundance levels of 0.10 kg/m^2; in locations where carcass abundance was less than 0.10 kg/m^2, enrichment index values averaged 0.19 ± 0.11(one standard error) as compared to 0.48 ± 0.13 in areas with carcass abundance above 0.10 kg/m^2. Carcass abundance of 0.10 kg/m^2 approximately equals 120 fish/km^2, above which marine N in juvenile coho salmon rapidly approached a 'saturation level'. Based on previous findings (Bilby et al. 1996, 1998), researchers in this study assumed that juvenile coho salmon were primarily incorporating marine N through direct consumption of salmon carcasses and eggs. Given this premise, the saturation level found in coho salmon parr could be interpreted as the maximum level of dietary enrichment for this trophic interaction. Based upon spawner escapement data and research findings, Bilby et al. (2001) conclude that the majority of coho salmon spawning streams in western Washington are well below capacity for incorporating more marine-derived nutrients.

From both a research and management perspective, there are numerous limitations to applying results from Bilby et al. (2001) as a standard for salmon escapement goals (many of which the researchers acknowledge). First, study sites were purposely chosen to only include areas with spawning coho salmon and no other returns of anadromous salmonid species. This implies that results may only be applicable in such areas and questions if marine nutrient dynamics would be similar in systems with returning runs of multiple salmon species. The temporal distribution of spawning by numerous species of salmon can mean prolonged input of marine nutrients, which may be more effectively incorporated within a system (due to nutrient flushing) at a lower density of spawners for a given species. Second, juvenile coho salmon alone are probably not an appropriate indicator for determining whether productivity in a system is nutrient limited (Simberloff 1998). The marine N signal found in juvenile coho salmon has been primarily attributed to direct consumption of salmon carcasses and eggs. If this is indeed the primary mechanism for nutrient uptake then isotope values from juvenile coho salmon are less revealing of other pathways for incorporation and trophic distribution of MDN within a system. Third, uncertainty remains as to whether increasing the input of salmon-derived nutrients to fluvial systems will subsequently result in higher returns of adult salmon. Results from the Bilby et al. (2001) study would suggest this due to higher $\delta^{15}N$ index values in juvenile coho salmon from systems with higher carcass densities. The effects of hatchery-origin salmon, that spawn naturally, must also be considered.

Gaps remain in our understanding of nutrient dynamics in fluvial systems. While it appears that salmon-derived nutrients can benefit sockeye salmon, cutthroat trout and coho salmon populations, at this time there are no research publications that directly establish the relationship between MDN and chinook salmon. ‘Ocean-type’ juvenile chinook, which comprise most of the production in Puget Sound, generally spend between three to nine months in freshwater before outmigrating (Healey 1991), a much shorter period than coho and steelhead (Montgomery et al. 1996; Healey 1991). Degraded spawning habitat and winter flow conditions, with direct influence on egg survival and emergence, may be more critical to chinook production than inputs of MDN. Upon outmigrating from the freshwater environment, juvenile chinook salmon may reside in estuarine environments for extended periods of time where conditions are critical for early growth and survival (Simenstad 1997; Simenstad et al. 1985).

Numerous questions arise in considering the potential role of MDN for ocean-type chinook salmon populations. Whether newly emerged chinook salmon fry actively feed on salmon carcasses and eggs has not been established and further questions if carcasses are retained for a sufficient period of time, especially in large river systems with peak winter flow events. The immediate benefits of MDN for chinook salmon fry is most likely limited given the relatively short time juveniles reside in freshwater. However, the River Continuum Concept (Vannote et al. 1980) suggests that upstream inputs of MDN affect downstream communities. This concept questions nutrient dynamics and source-sink effects within a river basin.

Ultimately, the benefits of MDN for juvenile chinook salmon may be more fully realized in estuaries (Simenstad 1997). That said, in some instances the eutrophication of estuaries associated with agricultural and urban development may be negatively affecting fish habitat and survival (Bricker et al. 1999). Currently, little is known about the effects of salmon and MDN on estuaries.

At a watershed scale, the connectivity of nutrient cycles and the pathways involved needs further investigation. Such considerations question the relative importance and actual contribution of MDN from different species of spawning salmon. In many river systems throughout the Pacific Northwest, returns of chum and pink salmon comprise the majority of spawner biomass. These species typically spawn in the lower portion of stream and river systems. This implies that chum and pink salmon contribute substantial inputs of MDN to environments used by ocean-type juvenile chinook salmon. Whether survival of juvenile chinook salmon is limited by nutrient deficiencies needs to be evaluated in a multi-species context. Furthermore, the relative contribution by adult returns of different salmon species to both ecosystem function and salmon populations with unique life-history strategies needs to be more fully recognized.

In considering the importance of MDN to ecosystem function and sustaining salmon populations, the large returns of adult salmon runs recently experienced throughout the Pacific Northwest dictates that an experiment is now in-progress. The current scenario provides unique research opportunities to assess if marine nutrient inputs are limiting salmon populations. This will necessitate that isotope methods are further developed and tested (see Kline 2002) to properly reveal MDN in food-web dynamics. Assessment of watershed nutrient levels will be necessary to determine regional variation. Identification of bottlenecks in survival to salmon populations will require careful monitoring of population dynamics across fish life-stages. Long-term studies on a larger spatial scale need to be initiated before we can properly understand the contributions of salmon and MDN to ecosystem function. The multiple values associated with salmon necessitates that this understanding be further developed and integrated between numerous disciplines before ecosystem based escapement goals for Pacific salmon can be a realized and effective management approach.

Appendix E. Escapement Estimation

Introduction

Accurate estimates of chinook spawning escapement are essential to management of Puget Sound chinook stocks. They represent the most immediate post-season monitoring of stock abundance and are essential to subsequent forecasting and reconstruction of cohort strength. Total escapement is also an invaluable measure for survival and productivity measurements, which is important in developing escapement goals and recovery objectives. With the availability of other relevant data, abundance reconstruction enables the estimation of cohort survival (returns per spawner), which, in turn, is the basis for setting harvest exploitation rate objectives. It is appropriate, therefore, to scrutinize the survey and computation methods utilized to estimate escapement with respect to the accuracy and precision of the resulting estimates.

The listing of the Puget Sound chinook has created further determination to improve escapement estimates. However, it is important to realize that accurate and precise estimates of escapement come at a cost. Given the limits on staff and funding, along with logistic limitations, a careful triage is required to determine where existing deficiencies should be addressed. The co-managers' chinook harvest management plan includes a mandate to insure effective monitoring of the productive status of Puget Sound chinook stocks.

There has not been a formal Puget Sound-wide review of escapement estimation methods since Smith and Castle (1994). However, a summary of escapement methods is documented each year, concurrently with preseason forecasts. A critical assessment of escapements has been a major task of the Chinook Technical Committee (CTC) of the Pacific Salmon Commission, especially those populations used as indicator stocks. Concerns about Puget Sound estimates has focused on the following issues:

1) accuracy and precision of estimates of total or partial escapement (including the testing of inherent assumptions);
2) Natural Management Units lacking estimates of total escapement;
3) currency of escapement goals: females or PED, vs total;
4) straying – contribution of hatchery-origin adults;
5) accounting of natural returns to hatchery rack;
6) age composition of escapement.

This document summarizes current methods for estimating escapement and describes recent work intended to validate or improve escapement estimates.

Current Methods

Spawner surveys, with the intent of estimating abundance, are conducted in all waters where naturally sustainable populations exists (category 1 and 2 watersheds). In addition, some category 3 watersheds are also surveyed. There are two basic types of surveys—census and index. Census surveys are conducted where all fish (carcasses or redds) can be counted. This implies that all redds and/or fish are visible and all spawning areas can be viewed so that there is no expansion of the estimate to account for unsurveyed areas. In the case of a redd census, all redds must be visible and all spawning areas must be viewed. In some areas, a marked redd census is used, where redds are marked, usually with a colored stone, to avoid recounting the redd during subsequent surveys.

Weirs can also provide opportunity to census returning fish. However, weirs are generally associated with the collection of hatchery brood stock and not natural spawning populations. In cases where excess fish are passed upstream, fish can be counted directly. Other situations include Baker Dam, which has a trap-and-haul facility to pass fish over the dam, as does the Mud Mountain Dam (Buckley Trap) on the White River. On the Snohomish system, chinook are trapped and hauled over Sunset Falls. Although counting sites such as these may provide accurate estimates of fish passing a single point, estimates may not necessarily reflect of spawning success.

With watershed that are too large to survey their entire length, and/or all potential spawning sites, index areas are used to estimate total spawner abundance. These are selected (non-random) sites where chinook are likely to concentrate. Although index areas may represent only a portion of the watershed, they usually incorporate a significant component of the spawning population. Index areas can be used to estimate either fish (carcasses or live fish) and/or redds. Surveys are conducted periodically throughout the spawning period, and include such information as location, time, date, water conditions, number of redds, live and dead counts, along with collecting scales for age data. Counts are conduct on foot or by floating the index areas. In the case of redd counts, aerial surveys are often used either exclusively or in conjunction with ground surveys.

Once the counts are completed and data assimilated, the actual estimates are usually calculated using peak counts, cumulative counts or area-under-the-curve (AUC). Peak count estimates are simply the highest number of observations made within a specific time period, such as one day. Once that number is identified it is expanded to account for such factors as non-surveyed areas, fish per redds, visibility, etc. Cumulative counts involve enumerating observed fish and/or redds over a period of time, usually the spawning period, and summing the observations. This usually requires some sort of marking program to prevent recounting. A more sophisticated variation of this is AUC which accounts for the entire duration of fish presence, using specific observation dates that are compared to the total spawning duration. This produces a curve of the counts that has typically been constructed for either redds or fish. This method has been widely used by many previous management biologists for various northeast Pacific salmon (Ames and Phinney 1977, Bue et al. 1998, Hilborn et al. 1999, Hill 1997, Liao 1994, Smith and Castle 1994). In the case of redds, the left side of the curve, the last date before the first redd is formed defines the beginning of the curve (i.e. the last date with zero redds). Ground observation and interpolation may be needed to specify this date. Straight lines are typically used to connect each subsequent count of visible redds, although some researchers have attempted curvilinear fits (Ames 1984). On the right side of the curve, the first date where the count is judged to be zero (known or interpolated from ground observation) forms the end of the curve. The area-under-the-curve (AUC) is the sum of the areas between each subsequent count, beginning and ending with the zero count dates, a method known as trapezoidal approximation (Hahn 1998, Hahn et al. 2001, Hilborn et al. 1999, Hill 1997). Each segment AUC is simply the sum of the two adjacent counts divided by two then multiplied by the number of days between the count dates plus one (i.e. simply subtract the earlier date from the later date). The total AUC is the sum of the segment AUCs. For redds, the primary variables are redd-life (the duration of redd visibility) and fish per female (since it is the female that builds the redd).

Nearly all escapement estimates of Puget Sound chinook are translated into total escapement for the watershed. The systems where escapement estimates reflect only the index areas are North Lake Washington tributaries and Skokomish River. Within the Lake Washington system, counts at the Ballard Locks estimate annual returns, but do not account for fall-back or pre-spawning mortality. Ballard counts also cannot be used to estimate escapement to individual watersheds. Skokomish mainstem counts are used to provide relative comparisons with two tributaries (Hunter and Vance creeks), which are generally not surveyed.

Improving current methods

There are four basic ways that may potentially improve escapement estimates: 1) expand indices (area of surveys), 2) conduct more frequent surveys, 3) re-establish base years by calibrating expansion factors or total estimates by comparing it with alternate methods, or by 4) testing basic assumptions such as expansion factors, spawner density, redd life, fish per female, adults per redd, etc.

Parameters such as confidence intervals and standard deviations have generally not been applied with any significance to escapement estimates. Exceptions include some of the work funded through the Chinook Technical Committee (CTC) of the Pacific Salmon Commission, such as those conducted on the Stillaguamish, Snohomish and Green rivers. Attention has focused on gaining more confidence of some basic assumptions, such as redd life and fish per redd. In many large river systems in Puget Sound chinook escapement is assessed by making repeated counts of redds, plotting these counts against time, then calculating the total number of redds from the area under the curve. Each redd has been assumed to represent one female and 1.5 males in calculating escapement. Whether made by aerial, boat, or foot survey, redd counts are subject to errors associated with visibility, insufficient survey frequency, observer error, false redds, superimposition, and the inability of distinguishing chinook redds from pink salmon redds. Assumptions regarding redd life and sex composition have been based on a few supporting, mostly old, studies, with the standard assumption for redd life as 21 days (Ames and Phinney 1997 and Orrell 1976 and 1977). Because the cumulative effects of these sources of error have not been quantified, the accuracy and precision of the resulting estimates is unknown.

A recent study (Hahn et al. 2001) examined redd estimators, as applied to chinook escapement to the Skagit and Stillaguamish rivers, and reached the following conclusions:

- The accuracy and precision of redd census ranged from very good (C.V. 10 – 15%) to uncertain, depending on conditions in each stream or river. Aerial surveys (particularly helicopter) were accurate in some streams, and varied from foot or boat surveys in others. More frequent aerial surveys were believed necessary to accurately define the spawning curve in some systems.
- The secondary assumption that females build only one redd was generally supported by field observations, though the potential for multiple redds per female or false redds exists in certain streams.
- Estimates of sex composition based on carcass counts or gillnet test fisheries engender significant, but unquantified bias. Thus the assumption that 1.5 males per female was not validated. Males and small chinook are undersampled by carcass surveys and gillnet samples.
- Intensive foot surveys to mark and monitor redds found that redd life varied significantly from 21 days in some systems.
- Covariance between the area under the curve and redd density is presumed, but should be quantified.
- Mark / recapture methods for estimating escapement and its variance, such as have been employed in the North Fork Stillaguamish River and Green River in recent years, are affected by several factors that bias their result. The resulting estimates (Conrad 1993, 1994, 1995, 1996, 1997; Nason 1999) were substantially lower than concurrent redd count-based estimates, and were probably affected by unequal probability of capture, non-random mixing and loss of marked carcasses from the study reach. However, recent

studies on the Green River show mark and release estimates to be higher than the standard redd and carcass estimates (Hahn et al. 2000).

Redd census techniques employed successfully in large river systems are usually supplemented by carcass counts and/or redd surveys in tributaries where aerial census may be impossible. Estimates of total escapement for a given stock may therefore be composed of several techniques. Details for each management unit are summarized within each watershed section.

CTC funded studies have specifically been devoted to improving estimates. On the Skagit attempts have been made to compare the existing escapement estimates with a live mark-recapture estimate. The primary objective of the study was to estimate the drainage-wide escapement of chinook salmon returning to the Skagit basin and to evaluate the fishwheel and beach seine sites in the lower Skagit River for capturing adult chinook salmon. The study was conducted for two years (2000 and 2001), and it was determined that these two methods alone would not capture enough fish to generate a reliable mark-recapture estimate of escapement (Smith et al, 2002). For 2002, the primary objective remains as a mark-recapture study. However, the planned method of capture included tangle nets and angling. In addition, radio-telemetry was also planned to investigate the distribution and behavior of chinook after capture and release.

Another mark-recapture study has also been underway on the Green River for three years (2000, 2001 and 2002). Adults are captured with a beach seine and released, with subsequent recapture within the spawning areas. This study has proved more successful than the Skagit study in that the number of marks and recaptures has been high enough to provide credible estimates. Studies have also been conducted on the Stillaguamish and Snohomish river systems. Final reports for all years should be forthcoming shortly

Oregon has used similar methods in assessing their coastal fall chinook populations. Standard index areas have been chosen based on survey history as well as being a valid representative of spawning escapement. which is indexed as the peak count of live and dead fish observed in a given survey area. Because standard survey sites were not chosen from a randomized sampling design, spawner density estimates obtained from these sites are used only to provide relative abundance (Jacobs 2001).

However, for coho Oregon uses a different approach. A review of the Oregon Coast Naturals (OCN) spawning survey program by Oregon State University Department of Statistics led to the initiation of the OCN escapement methodology study in 1990. This study involved the development and experimental implementation of a stratified random sampling (SRS) approach, which consists of randomly selecting spawning survey sites from geographical strata and estimating spawner abundance from visual counts in these survey sites (ibid). This approach follows EPA's Environmental Monitoring and Assessment Program (EMAP), which is similar to that of the National Park monitoring. The basis of this program is to avoid bias through random selection of sampling units and to use a sampling design that estimates population attributes that can produce reliable, absolute values of population abundance.

Some discussion has been initiated regarding its use for Washington chinook. However, there are several major disadvantages in implementing this sort of method. Among the most critical would be that present index areas would no longer be used, thus making past data unusable for comparison purposes. Because chinook spawn in specific areas, a large number of sampling sites would be required to provide adequate observations, and there would likely be many samples with no observations. The cost of identifying new sites and their subsequent monitoring would be more expensive and require additional staff to carry out than with current methods.

In general, assumptions regarding uniform spawning density have not been tested. This assumption applies not only to waters outside index areas but also to different times. Chinook will spawn in different areas in different years, depending upon changing environmental conditions, run size, human factors, etc., and the use of a single constant, or expansion factor, may not provide accurate estimates or be comparable from year to year. Survey conditions can also change, making it more or less difficult in observing fish and redds. In problem areas, estimates can be improved by expanding index areas. However, it should be noted that, in terms of recovery assessment, annual trends are as important as the escapement numbers, and changing survey procedures may result in estimates that are not comparable to previous surveys. In such cases, the importance of accurate estimates versus precise trend information must be weighed.

One remedy is to incorporate supplemental areas, which are spawning sites that are not included as index areas. Another method is to survey the entire watershed where chinook spawn. This is only feasible in smaller rivers where access is available throughout the entire length of the watershed or, in larger rivers, by using aerial-redd surveys where conditions allow complete view of the river substrate.

In summary, escapement estimates can be improved, but it is unlikely that there are new methods that will replace the current ones. Actual improvement of any population estimate will likely have unique requirements specific to the watershed. Some watersheds, for example, are inherently difficult to survey regardless of available resources. However, before a decision is made to invest resources to further improve an estimate, it is importance to weigh the needed information and the status of the stock against the potential benefits and costs..

Refining escapement goals

Fixed escapement goals have been used as the performance standard for harvest management. However, they were merely averages of escapements for various years during the 1960s and 70s (Ames et al. 1977) and did not necessarily reflect habitat productivity nor maximum sustain yield, upon which harvest goals were based. Because of the need to closely monitor the performance of the annual harvest regime, harvest management plans now calls for developing exploitation rate objectives for as many management units as possible, based on current and potential productivity. Basically this requires estimating the productivity (stock:recruit) function for the populations and implies that harvest rates can be associated with an escapement range for a given watershed.

Nevertheless, the question of escapement objectives remains under consideration within at least three forums. The Technical Recovery Team, which is coordinated through NMFS, has defined a number of parameters necessary for recovery. Among them is abundance of natural-origin recruits, which is expected to include both ESU and specific watershed criteria. The Ecosystem Diagnosis Treatment (EDT) process has also developed an initial review of some Puget Sound watersheds and identified escapement ranges based on properly functioning conditions (Molbrand 2000, Anonymous 2002). Finally the Chinook Technical Committee has been involved with a review of escapement goals throughout Washington (Hahn et al. 2001). All of the above review sources have started releasing results, and it is expected that additional information will be forthcoming. It is expected that escapement objectives will change as new information, such as habitat productivity, stray rates and other hatchery/wild interactions, become available.

The need to estimate escapement accurately is not lessened under this exploitation rate management system since escapement abundance remains a primary measure of stock health. If the harvest regime operates as planned, and abundance is close to what is forecasted, the escapement should also conform to pre-season expectations. The co-managers are committed to assessing the performance of the harvest regime annually, and modifying fishery regulations as necessary to assure that exploitation rate objectives are met. Over the longer term, regular assessment of stock productivity, for which accurate assessment of survival and productivity is essential, will also modify the harvest objectives to insure that recovery will not be hindered.

Straying

Estimating the contribution of first-generation, hatchery-origin adults to natural spawning is essential to understanding the natural productivity of any chinook population. Natural productivity (i.e. survival) can only be estimated by distinguishing hatchery and natural-origin components of harvest and escapement. In most Puget Sound systems, hatchery production is directed towards harvest augmentation, whereas only a few programs are directed at recovery. The concern is that hatchery fish may intermingle and interbreed with natural-origin chinook, resulting in direct interactions, such as competition for food and space and/or indirect interactions such as reduced fitness due to genetic modifications. Various studies with salmonids species have reported potential genetic and behavioral hazards to natural production caused by the interactions with hatchery fish. (Ames et al. 1984; Fleming and Gross 1995; Pearson and Hopley 1999; Reisenbichler 19??; Chilcote 2002).

Hatchery-origin adults are usually distinguished by some identifying mark, either externally, such as a fin clip (which may signify that the fish also carries a coded-wire tag), or internally, such as an otolith mark. Double index tagging (DIT) programs, which are intended to estimate mortality in selective fisheries of unmarked fish, involve coded-wire tagging two equal-size groups of hatchery releases, only one of which is externally marked by an adipose clip.

Estimation of stray rates is made more certain if hatchery production is mass-marked, which allows spent adults or carcasses to be quickly examined. Where DIT programs exist, unmarked fish will pass through an electronic tag detector to recover CWTed fish. Studies in the Green River suggest that carcass sampling provides superior estimates of the contribution of hatchery fish to natural spawning as compared to sampling extreme terminal (freshwater) catch. In the case of otoliths marks, otoliths are dissected from a sample of unmarked carcasses to establish the presence of this mark group. Otolith marking has been used successfully to estimate the stray rates of Tulalip Hatchery fall chinook into adjacent watersheds (Rawson et al. 2001).

In the case of recovery programs, it is not desirable to mark hatchery fish since they are liable to be harvested during selective fisheries. However, an internal or external mark (other than an adipose clip) would still allow the ability to identify hatchery returns in the escapement. This has been the case for Nooksack and White River spring chinook as well as for Dungeness River chinook. Selective fishing for chinook has not yet been widely implemented by the Washington co-managers, but mass marking programs have been initiated not just in anticipation of future selective recreational fisheries, but as a way to better determine hatchery/wild interactions and stray rates. In turn this will help address the productivity characteristics of the watershed.

Age and sex composition

Estimating spawning escapement and cohort reconstruction require information on the age and sex composition of the return. Escapement estimates, as discussed above, rest on assumptions about the number of redds that each female builds, and pre-spawning mortality. Reconstruction of the cohorts comprising brood year abundance requires estimates of the age composition of annual returns. The age and sex of returning adult chinook may be determined by sampling terminal or extreme terminal (i.e. freshwater) fisheries, carcasses of spawned-out fish, or fish returning to hatcheries.

Terminal fisheries, carcass surveys, hatchery rack collections are all used to obtain samples. However, each of these sampling methods may engender bias into the result. Gillnet gear that is designed to target chinook is often selective of larger fish, and may not catch jack males. The catchability of each size class of chinook may also vary under different conditions of flow and turbidity in the river. Terminal fishing occurring in the bays adjacent to the river mouth can be equally selective, and may intercept significant numbers of fish destined to other systems. Hahn et al. (2001) concluded that larger sample sizes from terminal fisheries would improve estimates. Recreational catch may also be selective, but it may be logistically difficult to obtain large enough sample sizes. In addition, recreational fisheries may not operate across the entire migration period nor target within terminal areas.

Carcass sampling tends to undersample small fish and males, but studies differ in their conclusions in this regard (Conrad 1996; various studies cited in Hahn et al. 2001). The magnitude of true bias is usually unknown, because carcass retrieval can only be compared with other, possibly biased, samples, such as those from fisheries or hatchery racks. The fieldwork involved is labor and time intensive, and frequently complicated by high flow, turbidity, and debris. ‘Carcass life’ (i.e. the time window available to sampling) is often affected by predators removing carcasses before they can be sampled, and by fish moving or being swept out of the sampling area. Carcass weirs have not been employed in Puget Sound streams.

Hatchery racks allow sampling throughout the entire migration period, allowing scales or other samples can be collected at frequent intervals. However, hatchery returns may not be representative of wild populations, particularly where non-indigenous stocks have been used. For many wild stocks there is no associated hatchery program, precluding rack and brood stock sampling. These include the South Fork Nooksack springs, Skagit falls (though broodstock collection for a PSC Indicator Stock has begun), Lake Washington / Cedar, and Mid-Hood Canal rivers.

In general, sampling should:

- encompass the entire migration period.
- be representative of single stocks or populations;
- Be designed to achieve unbiased and statistically significant results
- be random but represent the population.

Methods currently used for each management unit

Smith and Castle (1994) documented escapement estimate methods within Puget Sound and the Straits of Juan de Fuca. In general, these methods continue to apply. However, for most watersheds, there are on-going efforts to maintain and improve spawner estimates. The following reflects the current methods as of 2002.

Hoko: (Ground surveys, redd census)
The Makah Tribe and WDFW conduct surveys using cumulative redd counts for the mainstem and tributaries found between river miles 1.5 to 21.7, which represents the entire range where chinook spawn in the Hoko basin. Redd counts are multiplied by 2.5 adults/redd. There are ten mainstem reaches plus 13 reaches within tributaries, which include the Little Hoko River, a tributary to the lower mainstem, and Browne's, Herman, N.F. Herman, Ellis, Bear and Cub Creeks, which are tributaries to the upper mainstem. The Makah Tribe also surveys the mainstem and other independent tributaries in the Sekiu basin, including Carpenter, S. Fork Carpenter, and Sunnybrook Creeks, and unnamed tributaries (WRIA 19.0215, 19.0216, and 19.0218). The escapement estimates for these two rivers are based on total natural escapement for the Hoko basin, plus broodstock capture, and total escapement in the Sekiu basin.

Elwha: (Ground surveys, redd census using AUC)
Spawning chinook are limited to the lower 4.8 river miles below the dam. The preferred method of estimating adult escapement, in the mainstem, is plotting visible redds versus date and calculating the area under the curve, resulting in redd-days, which are divided by the 21-day redd life. The resulting redd total is added to the number of redds counted by the Lower Elwha Tribe in the 1 mile, Hunt's Road side channel index. The total redd count is then multiplied by 2.5 adults/redd.

Dungeness: (Ground surveys, redd index counts)
Since 1986, cumulative redd count surveys have been conducted from RM 0 to 18.7 in the mainstem Dungeness and from RM 0 to 5.0 in the Gray Wolf mainstem. Counts are multiplied by 2.5 adults/redd. A captive brood program has been underway in this system since 1992, with the first releases from this production effort occurring in 1995. The various families and year classes are uniquely marked with cwt and otoliths. Hence surveys also sample for these items.

Nooksack, North Fork: (Ground surveys, carcass index counts)
The primary difficulty is the turbid conditions that usually exist in the north fork, making redd counts impossible. Estimates are cumulative carcass counts in established index areas in the north and middle forks. Total estimate is scaled to a single year when carcass and redd counts were visible throughout the duration of the spawning period. With the return of otoliths marked fish, their sampling has become routine. Recent changes to production goal at Kendall Hatchery has led to the elimination of the summer/fall release program and reduction in the release of native, spring stock. Past escapement estimates have been complicated by spawn timing overlap of native and introduced stocks.

Nooksack, South Fork: (Aerial and ground surveys, redd census)
There are at least three groups of chinook that can be identified as spawning in the South Fork: 1) South Fork natives, identified by DNA and lack of other distinguishing marks, 2) North Fork natives as strays from the Kendall Creek hatchery restoration program (otolith marks, CWT) or natural strays (DNA) and 3) Green River /Soos Creek chinook as strays originating from hatchery programs past and present (DNA, adipose clips and CWTs). A total chinook estimate is derived from redd surveys conducted on foot by teams of two, done weekly from the middle of August

until the first week in November in all sections of the river and in 2.6 miles of tributary streams. Redds are counted, and expanded by a factor of 2.5 chinook per redd (i.e. 1 female and 1.5 males per redd) to obtain a total estimate. Because of high flows late in the survey season, the confidence in the total estimate deteriorates. Native chinook are estimated from the numbers of redds detected prior to September 29. An initial estimate of the North Fork native chinook is calculated from the proportions of carcasses which can be identified by otolith mark, or CWT and fin clip as coming from the recovery program. This estimate is subtracted from the total early native chinook estimate to provide an estimate of the South Fork native chinook spawning population.

Samish: (Ground surveys, redd/carcass census)
This system is considered a Category 3 watershed, which, historically, did not possess as sustainable chinook population. However, large numbers of summer/fall chinook (introduced) fish are released from Samish Hatchery each year. As a result, natural spawning does occur in the river below the hatchery. In addition, fish surplus to hatchery needs are released above the hatchery. This stock is managed for harvest augmentation and is managed only for achieving hatchery brood needs. Estimates are made using peak visible redd counts, multiplied by 0.95 to estimate true redds and then by 2.5 fish per redd. If river conditions are not conducive for redd counts; carcass counts are made on weekly basis. Fish spawning above the hatchery are counted as they are passed upstream over the rack.

Skagit: (Mainstem-aerial surveys, redd index counts; tributaries-ground surveys, redd census and index counts)
The entire Skagit and known spawning areas in the Sauk and Cascade rivers have been surveyed by helicopter on either a weekly (odd years) or biweekly (even years) basis. During odd years, surveys are concentrated within the first half of the run with a straight line connecting the peak to the end of redd visibility. This is due to the large numbers of pink salmon spawning in the same location as chinook salmon. Earlier chinook spawners are located in the upper Sauk, Suiattle and Cascade rivers. Later spawners typically spawn in the mainstem Skagit, associated tributaries and the Sauk River.

For the earlier-timed chinook, data from 1994 to present is not comparable to previous escapement estimates. This is due to a new escapement methodology, using expanded cumulative redd counts, which is thought to represent the total spawner population better than the pre-1994 method using peak live plus dead counts. (Rebecca Bernard, Skagit System Co-op, personal communication).

Studied funded through the Chinook Technical Committee (CTC) has provided initial assessments of the validity of the current escapement estimates. Work conducted in 1998 and 1999 showed that the 21-day redd life was a valid assumption for Skagit chinook (Hahn et al. 1998) But work still remains in testing the 2.5 fish per redd. To accomplish this, and to establish as base year for future estimates, the basic plan was to proceed with a mark and recapture study, using a fish wheel to capture adult chinook. This fish wheel was used for two years without success (too few fish were captured). In 2002 attempts were be made to use a combination of collection methods including tangle nets, angling and radio-telemetry (CTC January 8, 2002).

***Lower Skagit Mainstem fall*:** Data are total escapement estimates based on redd counts from the mainstem Skagit between the town of Sedro Woolley and the mouth of the Sauk River and in Finney and Day creeks. Three fixed wing aerial surveys are conducted from RM 15.6 to RM 67.1. There is a turbidity problem downstream of the Sauk, which questions the assumption of old surveys of 100% visibility. AUC estimates for three reaches using Sept 15 as start date on lower reach and Sept 1 for upper two reaches. End dates are December 1 for lower and middle reach and Nov 15 for upper reach. The old method used Sept 1 - Dec 1 for all reaches. Tributary census is conducted in Finney, Johnson, Jackson creeks.

Upper Skagit Mainstem/Tributaries *:*This stock was formerly known as Upper Skagit Mainstem/Tribs summer chinook. In the 2002 SaSI revision, the run-timing designation ("summer") has been dropped from most Puget Sound chinook stock names because timing designations have been applied inconsistently to Puget Sound chinook stocks. Total escapement estimates are based on redd counts from the mouth of the Sauk River to Newhalem, the lower Cascade River (RM 0.0 to 6.5) and in Illabot, Diobsud, Bacon, Falls and Goodell creeks. Surveys include three helicopter flights of upper mainstem, plus two helicopter flights and three ground surveys on the lower Cascade (RM 0.0 – 0.9), using Aug 15 to Nov 1 as AUC period (previous assumption has been Nov 8).

Lower Sauk (fall**):** Total escapement estimates are based on redd counts from the mouth of the Sauk upstream to the town of Darrington (RM 0.0 to 21.1). Aerial counts below mouth of Suiattle are not conducted due to turbidity. This sediment concentration is believed to inhibit spawning downstream, and past estimates assumed 22% of redds occur below RM 13.2. However, a simulation based on 1996 flights suggested that the majority of fish spawn below RM 13.2. Three flights are made above confluence (RM 13.2 – 21.1 Darrington Br.), with foot surveys of Dan Creek slough, which is now part of the mainstem. The estimate is a redd census above RM 13.2 plus assumed number downstream plus tributary counts times 2.5 fish per female.

Upper Sauk spring: Total escapement estimate is based on redd counts from the town of Darrington up to the forks (RM 21.2 to 39.7), in the North Fork Sauk from the mouth upstream to the falls and in the South Fork Sauk from the mouth to about RM 2.5. A new escapement methodology was developed beginning in 1994, using expanded cumulative redd counts, which are thought to represent the total spawner population better than peak live-plus-dead counts. (Rebecca Bernard, Skagit System Co-op, personal communication). The new estimates are not comparable to the estimates in the 1992 SASSI.

Surveys include five helicopter surveys and six ground surveys to monitor redds and count carcasses. Foot 'census' is thought to underestimate numbers due to width and depth of some reaches, and the fact that foot counts consistently yield lower numbers than aerial counts. Aerial-based AUC determined endpoints of Aug 15 and Nov 1. Redd life arbitrarily assumed to be mean of values derived from foot survey (22.9 days) and back-calculation from aerial AUC (37.5 days) = 30.2 days. Total escapement is based on 2.5 fish per redd. Other samples have show different female to male ratios such as the lower river test fishery (1.65) and carcass surveys (1.42).

Suiattle: Total escapement estimates are based on redd counts in Big, Tenas, Straight, Circle, Buck, Lime, Downey, Sulphur, Milk creeks. As mentioned above, new escapement methodology was developed beginning in 1994. Prior to 1994 four index areas (Big, Tenas, Buck, Sulphur) were used, averaging peak live-plud-dead count/mile from these areas. Since 1994 cumulative redd counts have been used. Index areas now include Big, Buck (excluded summer strays – early Oct), Circle, Downey, Lime, Milk, Straight, Sulphur and Tenas creeks along with Whitechuck River. The estimate assumed no redds in the turbid portion of the mainstem. Of all systems in this study, Siuattle thought to have highest potential for multiple redds per female. However, the present estimate remains based on 1 female per redd, or 2.5 fish per redd.

Upper Cascade springs: Total escapement estimate for this stock is based on redd counts from the mainstem Cascade River above RM 7.8, the lower reaches of the north and south forks of the

Cascade, and in Marble, Found, Kindy, and Sonny Boy creeks. As with the other early stock, new escapement methodology was developed beginning in 1992. Data for the estimates originated from five surveys conducted on foot and two helicopter flights (RM 7.8 – 18.6). Redds are multiplied by 2.5 fish per redd.

Stillaguamish: (Ground and aerial surveys, redd census using AUC (NF) and peak counts (SF))
Smith and Castle 1994 mentioned that the Stillaguamish escapement estimate used the same method as Skagit (aerial survey calibrated by foot surveys of index reaches). One to three flights have been used, with assumed starting dates for redd visibility. Redd counts were summed at 21-day intervals to get cumulative total redds times 2.5 fish per redd. Studies began in 1998 to improve the accuracy and precision spawning estimates by testing redd life and the number of female per redd. Aerial surveys were increased as well as the foot surveys, and both were compared throughout the sampling period.

North Fork Stillaguamish summer: Escapement estimates are made using cumulative redd counts within the mainstem and North Fork derived by graphing visible redds versus survey date.Although there were some discrepancies between redd count on the foot versus floot surveys, Hahn (2001) concluded that the estimates of chinook redds and of female spawners were precise and accurate. Seventy-five percent of the redds were censused with surveys every three to five days; water remained low and clear during this time with little canopy overhang, and good estimates of redd life were made (20-day).

South Fork Stillaguamish fall Escapement estimates are based on peak redd counts multiplied by 2.5 fish/redd. Tributaries surveyed include Boulder, Squire and Jim creeks. Assumption include: zero redds below the confluence of the North and South forks, 2.5 fish per redd and 21-day redd life. Hahn et al. (2001) stated precision and accuracy of the fall chinook estimate was uncertain. The primary problem in the AUC method was due to the inability to measure redd life. Low redd density and poor visibility at times also attribute to this uncertainty.

Snohomish River: (Aerial and ground surveys, redd census using AUC; direct census for Sunset Falls, index on Sultan)
Skykomish This stock now includes Snohomish summer, Wallace Summer and Bridal Vail Creek fall chinook stocks as well as a portion of the Snohomish fall chinook stock. Spawning occurs throughout the mainstem Skykomish and Snohomish rivers, Wallace River, Bridal Vail Creek Sultan River, Elwell Creek and in the North and South Fork Skykomish including fish passed above Sunset Falls. Natural spawning also occurs in the Wallace River, but many of these spawners originate from the Wallace River Hatchery, located at the confluence of May Creek and Wallace River. Escapement estimates are derived using cumulative redd curves from aerial surveys in index area RM 20.5-49.6 on Skykomish mainstem and South Fork to Sunset Falls. Calculation uses 21-day intervals. Additional surveys are conducted on Wallace River using cumulative redd counts times 2.5 fish/redd and .95 (true redds). Estimate is based on mid-Sept visible redds / total escapement ratio in prior year. Added to this is the number of fish trucked above Sunset Falls.

Snoqualmie*:* The Snoqualmie stock is composed of Snohomish fall chinook, which spawn in the Snoqualmie River and its tributaries, including Tolt and Raging rivers and Tokul Creek. Spawning also takes place in Pilchuck and Sultan rivers. Spawn timing occurs from mid-September through October. Snoqualmie escapement is based on aerial survey of 10.1 miles of index out of 39.6 miles of river below Snoqualmie Falls, and calculated using area under the curve. Redd days are divided by 21-day redd life times 0.95 and 2.5 fish per redd. No expansion factor is used.

Both sets of estimates are intended to be total estimates although there are some small tributaries that are not surveyed nor included in the final estimate. However, it is considered to be less than five percent of the surveyed areas.

Cedar River: (Ground surveys, live counts using AUC)
Cedar River escapement is estimated using live counts, plotting counts versus survey dates and calculating the area under the curve. Counts are obtained from float surveys throughout the river length below the dam. Redds have been enumerated since 1999, and at some point redd counts may be used to produce escapement estimates.

North Tributaries: (Ground surveys, live counts in index areas using AUC):
Spawning ground index areas have been established in Bear and Cottage creeks. Since 1998 other portions of the Bear Creek watershed are also surveyed annually, but are not part of the index areas used for estimates. There is no expansion to unsurveyed areas in other north tributaries. Escapement for Bear and Cottage creeks is based on live counts and area under the curve methodology. The index areas are: Bear Ck--RM 1.3 to 8.8, Cottage Lake Ck.-- RM 0-2.3.

Issaquah Creek: (Ground surveys, carcass and live fish counts using AUC):
This watershed is not believed to have historically supported a sustainable population of chinook and is classified as a Category 3 system. Returns to Issaquah Creek are believed to be entirely the result of hatchery production. Many more fish return beyond brood stock needs and the surplus is allowed to spawn naturally. Escapement estimates on Issaquah Creek are calculated as the sum of the individual carcass counts plus the live count from the last survey. For the East Fork, the estimate is based on live counts and area under the curve methodology.

Green River: (Aerial and ground surveys, redd index counts)
There are a considerable number of hatchery fish released from this watershed each year, and, as a result, the proportion of hatchery strays among natural spawners is high. Based upon CWT recoveries from carcasses sampled on the spawning grounds, the estimated annual proportion of hatchery strays averages about 60 percent, and ranges from about 25 to over 90 percent of the total natural spawners.

The standard method used to estimate the annual natural spawning escapement in the system employs the use of a single 1.6 mile index reach (River Mile 41.4 to 43.0) where individual redds are counted and marked weekly by raft to obtain a season cumulative redd count. Concurrent weekly aerial counts of visible redds are made in all reaches (including the index reach) from RM 29.7 to 47.0. At the end of the spawning season, the highest (peak) weekly aerial count of visible redds in the index reach is compared to the cumulative total of redds in the index reach, and an adjustment factor is derived. The peak weekly aerial count from non-index reaches is adjusted by this factor, and an estimate of cumulative redds is obtained for the reaches surveyed only by air. This estimate, when combined with the cumulative redds in the index, yields the total estimated redds for the surveyed portion of the mainstem Green.

An expansion factor of 2.6 is then applied to the surveyed mainstem redds to estimate the total redds for the entire system, including tributaries. This expansion factor was derived by Ames and Phinney (1977) after comparing their estimates of escapement in the surveyed reaches in 1976 and 1977 to estimates of total escapement in the system obtained from independent mark-recapture studies conducted by the Muckleshoot Tribe and the U.S. Fish and Wildlife Service in those years. Total system redds are multiplied by 2.5 fish/redd to convert system redds to the escapement estimate of individual chinook.

Beginning in 1999, funding originating from the Pacific Salmon Commission has been directed at improving spawning estimates on the Green River. Objectives have included estimating population size using live mark and recapture, developing new redd index expansion, comparing area under the curve method, testing chinook redd visibility, estimating number and proportion of hatchery-origin chinook and age composition. This work continues through 2002.

Puyallup (fall): Ground surveys, cumulative redd counts (even years), AUC (odd years)

With the large hatchery releases into Puyallup River, it is likely that some unquantified proportion of natural spawning fish are hatchery origin. Thus the extent of natural sustainability is unknown. Puyallup basin hatchery chinook production is currently 100% adipose marked, which will help determine natural production levels and stock status.

Annual spawning ground surveys are reliable in the South Prairie Creek system (considered to be the most productive portion of the watershed) and in the mainstem tributaries, where fish and redds are observable. In other spawning areas (Puyallup mainstem and the Carbon River), glacial flour reduces visibility and prevents credible observation in most years. Historically, estimates were based on the 1975 and 1976 tagging studies, which used South Prairie Creek index peak live count multiplied by a factor of 37 to estimate total escapement. However, there has been a lack of confidence in this method, and beginning in 1999 estimates were calculated using a different method. This involved using South Prairie Creek cumulative redd counts during even years, while odd years would be based on area under the curve (AUC) using live counts. This difference was needed to adjust for the presence of pink salmon during odd years. Redd based estimates can also be calculated for the following Puyallup River tributaries: Fennel, Canyon, Kapowsin and Clarks creeks. In 2000, the tributary escapement ratio was applied to the mainstem Puyallup to estimate Year 2000 spawners. For the Carbon, in 1999 water conditions were conducive for good redd counts within some river reaches. Reaches with incomplete data were expanded using South Prairie Creek spawn timing-curve. In 2000, river conditions did not allow counts, and an indirect estimate of relative returns between 1999 and 2000 were used. Although this method is considered an improvement over the old method, escapement estimates previous to 1999 are not comparable to recent year estimates. .

White River Spring Chinook: (Trap census over dam, no estimate below dam)
Although there has been a significant increase in the number of chinook returning to the White River, it is largely due to the successful hatchery program. There is no evidence that the population has re-established itself naturally or achieved self-sustainability. Improvements have been made in the upper watershed related to habitat and fish passage, but those actions have not been necessarily credited with the increased abundance levels. There is also concern that the increased numbers of chinook are, at least partially, attributable to a fall stock that has become more predominate. Recent year spawning information shows that the fall run of chinook has increased in abundance. However there has been no estimate of total escapement. Those fish passed over the dam are counted, but fish spawning below the dam are not surveyed. However, chinook are enumerated in Boise Creek and the lower White River below Buckley Trap.

Nisqually: (Ground surveys, fish and redd index, peak counts)
Given that a large number of hatchery fish are released into this watershed, it is believed that a significant proportion of natural spawners are hatchery strays, but no direct information is

available to verify this. This system is difficult to survey since it is glacial fed. Abundance estimates are fair at best; stock origin information is poor.

Since 2000, all hatchery chinook have been marked, making it possible to determine the hatchery/wild composition of natural chinook spawners in the future. Spawning surveys are conducted on Nisqually mainstem from RM 21.8 to 26.2 and on Mashel from RM 0 to 3.2 to obtain peak redd count on the Nisqually and peak fish count of the Mashel. An expansion factor of 2.5 is used for the Nisqually relative to the Mashel, followed by a 6.82 expansion for both systems. Ohop Creek (RM 4.6-6.3) has also been surveyed for cumulative redd counts and carcass sampling the last two years (2001 and 2002).

Skokomish: (Ground counts, fish and cumulative redd counts in index areas)

As described in the current co-managers' Puget Sound Comprehensive Chinook Management Plan, the immediate and short-term objective is to manage Skokomish River chinook salmon as a composite population, comprised of naturally and artificially produced chinook. Hence, natural production is dependent on the chinook hatchery program to partly support natural production. Based on the sampling of adult chinook carcasses on the natural spawning grounds, chinook released from the George Adams Hatchery on Purdy Creek or from Endicott Ponds on the lower Skokomish River stray in substantial numbers onto Skokomish system natural spawning areas. Hatchery chinook releases are not currently mass-marked, but they are now double-index tag groups. In addition, genetic (allozyme) analysis results to date suggest that there is no significant genetic differentiation between Skokomish natural spawners and George Adams hatchery chinook (A. Marshall, WDFW memo dated May 31, 2000).

Chinook spawning takes place in the mainstem Skokomish River up to the confluence with the South and North Forks at RM 9, in the South Fork (primarily up to RM 5.5), and in the North Fork from RM 9 to 17 (where Cushman Dam blocks further access). Natural escapement estimates are based on counts of chinook redds in index areas in the mainstem Skokomish (RM 2.2 to 9.0), North Fork (R.M. 9.0 to 12.7), and South Fork (R.M. 0 to 2.2). In addition, escapement estimates are made for tributaries including Purdy Creek, Vance Creek, and Hunter Creek.

Since 1991, live and dead adults, along with visible redds were counted in Skokomish River index areas using foot and raft surveys (Smith and Castle 1994). Surveys were done every 10 to 14 days from late August through October. In one index area of the Skokomish (RM 8 to 9), new redds were flagged and visible redds were counted each survey, cumulative redds for the season was determined, and escapement for this index was estimated as cumulative redds times 2.5 adults/redd. For each remaining section, the peak count of visible redds in a section was multiplied by the ratio in the RM 8 to 9 index of cumulative redds :: number of visible redds at peak which was then multiplied by 2.5 adults/redd to estimate escapement for a section.

Since 1991, escapements to Hunter Creek and Vance Creek were estimated using the spawners/mile for RM 0.8 to 2.2 in the South Fork and the available habitat in each creek (i.e., 1.7 miles for Hunter Creek and 0.5 miles for Vance Creek). Escapements to Purdy Creek were based on the counts of live chinook downstream of George Adams Hatchery (Smith and Castle 1994).

To improve escapement estimates, (1) surveys were scheduled every 7 to 10 days beginning in 1998, (2) new redds and visible redds were counted each survey in more sections of the mainstem Skokomish (RM 5.3 to 6.3, 6.3 to 8, and 8 to 9) and South Fork (RM 0 to 2.2) beginning in 2000, (3) a helicopter flight was made most seasons during peak spawning to count redds and adult chinook in the South Fork upstream of RM 2.2, and (4) foot surveys were made in Hunter and Vance creeks to spot check chinook abundance and better determine escapement there.

Coded-wire tag (CWT) data and age and sex composition data have been routinely collected for chinook returning to George Adams Hatchery. More intensive sampling has been done since 1998 on the natural spawning grounds; however, more frequent sampling would improve sample sizes. The mass marking of chinook released from the hatcheries would improve the ability to determine both the level of straying by hatchery chinook and natural chinook productivity in the Skokomish River system.

Mid-Hood Canal: (Ground surveys, live peak fish counts in index areas)
The Mid Hood Canal management unit is comprised of chinook populations of the Hamma Hamma, Duckabush, and Dosewallips watersheds. All of these populations are at low abundance. As described in Smith and Castle (1994), chinook escapement for the Hamma Hamma, Duckabush and Dosewallips rivers was estimated as (peak count of live fish in each stream) x (escapement for Skokomish RM 8-9 index / peak live count for Skokomish RM 8-9 index) x (available habitat / surveyed habitat in each stream). This method was used since few chinook adults or redds were counted and chinook spawner surveys were limited to the lower reaches of each stream.

In the Hamma Hamma River, most of the chinook spawning area is currently being surveyed. A cooperative supplementation program was initiated in 1995 to rebuild chinook abundance. Since 1998, abundance has increased and escapement was estimated from counts of live chinook using the area-under-the curve (AUC) method.

In the Dosewallips and Duckabush rivers, the reaches surveyed are spawning and transit areas, but do not include all spawning areas. Upper reaches have been occasionally surveyed in the Dosewallips and Duckabush since 1998, but few adults have been observed. It has been possible to count chinook redds in the upper Dosewallips and Duckabush river reaches (especially in years without pink salmon). However, counts of live chinook are conducted on in the lower reaches since chinook redds cannot be identified due to concurrent spawning of summer chum salmon. Current escapement estimates are derived from counts of live chinook adults and chinook redds.

It has been assumed that many of the naturally-spawning chinook in the Hamma Hamma, Duckabush, and Dosewallips rivers have, in recent years, been due to straying of hatchery spawners as well as adult returns from hatchery fry released into these rivers. However, sampling for CWTs and age information indicate that few hatchery adults have been recovered. The mass marking of chinook released from the hatcheries would improve the ability to determine both the level of straying by hatchery chinook and natural chinook productivity in these rivers. In addition, a smolt trap was installed on the Hamma Hamma River in 2002 with one objective being to assess natural chinook productivity.

Priorities for Improving Escapement Estimation

To identify priorities for improving escapement estimates, recovery goals and objectives must be clearly stated. The basic template should refer to the ESU as a whole rather than individual stocks. Since recovery can represent any number of different outcomes, the process must be iterative and based on the outcomes of strategies that may be experimental. However, regardless of the specific results, the basic guidelines of a healthy ESU can be stated.

Populations have been classified according to the historical presence of chinook and the present status of native (indigenous) stocks. Category 1 watersheds are those that possess indigenous stocks; Category 2 are those that once possessed sustainable indigenous chinook populations but they have either been lost or no longer sustainable; Category 3 watersheds are those that historically never possessed sustainable populations of chinook.

Category 1 watersheds would be of high priority, as would those in Category 2. Within the first category, highest priority would go to those stocks that are at critical abundance levels and where escapement estimates are considered unreliable (imprecise and inaccurate). Perhaps the single stock that best fits this would be the South Fork Nooksack stock. Another concern would be White River spring chinook. Both of these populations have been recently infiltrated with other stocks, which is causing some concern regarding genetic integrity in the direction of recovery. Cedar River chinook is another population that needs close scrutiny. Although the escapement greatly improved in 2001, previous years returns were in dramatic decline, with the 2000 estimate of 120 adults. For other systems like the Skagit, Stillaguamish and Snohomish, as mentioned, additional studies have been underway to test some of the major assumptions, and it is believed that this will improve accuracy and precision of current methods. In the Green River , a mark and recapture estimation method has provided significantly different results than the traditional method. Analysis of the differing escapement estimates for 2001 and 2002 will help determine the method used in future An important component on the Green is determining stray rates. Since all hatchery fish are now been marked before release, the estimation natural-origin recruits and habitat productivity will improve.

As important as accurate escapement estimates is the need to identify hatchery stray from natural-origin recruits. This is especially true for Category 2 watersheds where past management direction has focused on hatchery production at the expense of natural sustainability. For Nisqually and Puyallup chinook, marking of hatchery fish and subsequent evaluation of natural production must be maintained as an important objective. One difficulty common to both of these systems is inability to survey mainstem spawning reaches because of glacial turbidity. Experimental application of the “change in ratio” method, which estimates total natural escapement and the proportion of natural-orogin adults, began in 2001

Past management for Skokomish River has also been hatchery-oriented, and to date there has been no attempt to determine stray rates and natural productivity. It would also be useful to test the assumptions for Vance and Hunter creeks, which are estimated indirectly. A production study on the Hamma Hamma is currently underway that involves intensive spawner surveys as well as smolt out-migration

Appendix F. Selective Effects of Fishing

Introduction

The direct juvenescence or 'fishing-down' effect (shift toward younger ages and smaller fish) that must result from size-selective fishery harvest has been recognized for nearly 100 years (see Ricker's (1975, p. 260) discussion of Baranov's 1918 paper). But it seems only very recently that the possible genetic impacts of selective fisheries on fish populations have generated widespread concern among fishery scientists and ecologists. For example, Conover and Munch (2002) published a highly visible article noting that "current models and management plans for sustainable yield ignore the Darwinian consequences of selective harvest." In a similar vein, in the leading European quantitative fisheries journal, Law (2000) noted that "Fisheries managers should be alert to the evolutionary changes caused by fishing, because such changes are likely to be hard to reverse ...." Although this general concern may appear to be very recent, astute fisheries scientists have long speculated concerning the possible genetic impacts of selective fisheries on chinook salmon populations. Indeed, nearly 100 years ago Rutter (1904) expressed concern that gillnet fisheries in California's Sacramento River, selective for larger and older chinook salmon, might generate long-term selection toward age two male jacks and small adults due to selection against survival and reproduction of larger and older adults. More recently, but still a full thirty years before the recent Conover and Munch paper, Ricker (1980, 1981) published extremely provocative reports concerning the possibility that size-selective fisheries on chinook salmon might, in the long-term, result in age composition of chinook salmon populations that would be composed almost exclusively by age 2 male jacks and age 3 adult females. Thus, it is accurate to state that the potential long-term consequences of selective fisheries on chinook salmon have been recognized for almost 100 years. Yet, it is also accurate to state that fishery management plans have not yet attempted to address these potential long-term consequences. In part this is because much of the evidence for selective effects of fishing (e.g., change in the size or age composition of catch or spawners) is circumstantial, and is strongly influenced by other factors such as marine productivity.

Selective Fisheries

It is important to define more explicitly and carefully a number of terms and concepts. In particular, it is critical to define carefully just what one means by "selective fishing", to distinguish among the kinds of selective fishing to which chinook salmon populations may be exposed, and finally to distinguish between the rather immediate and direct fishing-down consequences of selective fishing and the potential long-term genetic consequences of selective fishing.

Generally, a fishery is characterized as selective whenever different components of a population of fish are exploited at different rates in recreational or commercial fisheries. Traditionally, most fisheries have been sex-selective (e.g., only males may be harvested in the commercial fishery for Dungeness crabs, Cancer magister) and/or size-selective (e.g., groundfish fisheries in which regulated codend mesh size theoretically allows small fish to escape whereas large fish are trapped in the codend; or the minimum size limit for male Dungeness crabs). In fisheries for chinook salmon, there are no sex-selective fisheries of which we are aware, but most fisheries are size-selective. For example, ocean commercial and recreational fisheries typically have minimum size limits, thereby generating greater exploitation rates on larger and older fish than on younger and smaller fish. Terminal gillnet fisheries typically select for fish that are within an intermediate size range that usually dominates runs. Often, such terminal gillnet selection is almost *"age-selective"* fishing. For example, in California's Klamath River the Native American gillnet fishery uses a mesh size that deliberately targets age 4 fish; most age 3 and younger fish pass through nets whereas many age 5 fish are too large to be caught by gill nets.

The above examples of selective fisheries apply within individuals populations of fish. Other types of selective fisheries operate in the peculiar context of ocean and freshwater fisheries for salmon. First, in both ocean and terminal fisheries, salmon managers must grapple with the so-called "mixed stock" harvest problem (see, e.g., Bevan 1987). In the ocean, a large number of salmon stocks originating from different river basins may be vulnerable to fishing at similar times and locations and may therefore suffer similar ocean exploitation rates. Optimal harvest policies would instead call for application of *stock-specific* exploitation rates that depend on the underlying stock productivity which, of course, must vary among salmon stocks. For a variety of reasons, the time, location or physical attributes of fish that may be caught in ocean fisheries may be deliberately structured so as to be *stock-selective.* For example, ocean fisheries off California and Oregon are structured so that the overall ocean exploitation rate on Klamath River fall chinook is quite low (to allow for terminal harvest in recreational and Indian fisheries), whereas ocean exploitation rates for chinook salmon originating from the Sacramento River (with no Indian terminal fisheries) are much higher. Mixed-stock fisheries are often constrained so that the exploitation rate appropriate to commingled weak stocks is not exceeded.

Similar, but often unintentional, *stock-selective* fisheries may take place in freshwater as a consequence of regulations. For example, in a large river system with a large number of distinct chinook salmon stocks, each with its own distinct river entry pattern, open and closed periods for fisheries may result in differential exploitation rates being applied to different stocks. If harvest in not allowed until a substantial number of fish have escaped to spawn, then it seems inevitable that exploitation rates are lower for those stocks that enter earlier as compared to those stocks that enter when fisheries are open. The most extreme examples of stock-selective fisheries for chinook salmon are those that call for the release of all fish with adipose fins present clips, whereas a certain number of fish (specified by bag or possession limits) may be retained so long as adipose fins are not present. These policies are deliberately designed to produce, at least in theory, greater exploitation rates for hatchery fish (often marked) than for wild fish (typically unmarked). Finally, ocean fisheries may also be *species-selective* as, for example, results when coho salmon must be released if caught whereas chinook salmon may be retained.

The "fishing-down" process and long-term genetic selection

The "theory of a fishery", as first advanced by Baranov (1918; see Ricker 1978), recognized *fi*shing-down as an inevitable consequence of size-selective fishing when only fish above a certain minimum size limit were legal targets of exploitation. The direct cumulative effect of removing larger and older fish is to shift the age structure of a fish population toward younger and smaller fish. Although these historical results were obtained for typical iteroparous (repeat spawning) teleost fish, similar results obtain for a semelparous (single spawning) chinook salmon population subjected to a size-selective ocean fishery (Hankin and Healey 1986). In classical fisheries population models, growth rates of fish are fixed and independent of population density, and fishing down-effects are therefore predictable and reversible. The extent to which genotypes of a populations are changed by selective fishing must be related to the harvest rates imposed by these fisheries and their duration. If selective fishing were eliminated, then one would expect the age and size structure of a population to return to exactly the state that existed prior to introduction of size-selective fishing. (Possible to make a general statement that selective effect is dependent on the harvest or exploitation rate, so that reducing the rate would reduce the effect?)

Concerns regarding the potential genetic impact of fishing have arisen in part because minimum size limits theoretically result in differential exploitation rates being applied to *fast-growing* as opposed to *slow-growing* fish. If growth rates of fish were genetically inherited and if realized size at age were highly correlated with genetically inherited growth rates, then the greater mortality on fast-growing fish and resulting dominance of slow-growing fish among spawners would, over the long-term, result in selection for slow-growing fish.. If such fishery-induced genetic changes took place, then a population would not return to its original state if fishing were eliminated entirely. Instead, if fishing were relaxed or eliminated slow-growing fish could become the norm. Exactly this kind of selective fishery result was documented, under a controlled laboratory setting, in *Menidia menidia* by Conover and Munch (2002). These laboratory results may or may not be relevant to "real" fish populations and fisheries, however.

Long-term genetic changes due to selective fisheries

***Size-Selective Fisheries*.**

In ocean fisheries for chinook salmon, minimum commercial size limits typically mean that only a fraction of the age 3 adults from a given stock are vulnerable to commercial capture. If those age 3 fish that are above the legal size limit were genetically programmed "fast-growing" fish, then one might imagine that selective fisheries would be generating long-term selection for reduced growth rates, as described above.

Possible fishery-induced selection for reduced growth rates would, however, be complicated by several factors in chinook salmon fisheries. First, the actual size that a salmon reaches at a particular age may not be highly correlated with a genetically determined "growth rate" for several reasons. The realized size of a fish at a given age must reflect unknown interactions between inherent growth rate, variability in supply and quality of food, and variability in environment (especially variability in water temperature). Actual size at age may not, in general, be highly correlated with some underlying "growth rate"

Second, long-term genetic selection due to size-selective ocean fisheries may be stronger for (reduced) age at maturity than for growth rate. As shown by Hankin et al. (1993) and others, age at maturity is an inherited trait in chinook salmon. Generally, older aged parents will produce progeny that mature at older ages, whereas younger aged parents will produce progeny that mature at younger ages. This kind of effect is especially pronounced for age 2 males (jacks). If jacks are used as parents, there will be a strong tendency for male progeny to also mature as jacks. Therefore, if younger aged salmon spawned randomly on the spawning grounds, then size-selective fisheries for chinook might select for earlier age at maturity.

Third, for chinook salmon (see Hankin 1993 and references therein) there is substantial evidence that age at maturity depend in part on size at age. For a fixed age, say age 2, fish that are smaller are less likely to mature at that age than are fish that are larger. Through this interaction between size at age and maturity, size-selective fisheries, through removal of fish that are larger at age, might instead select for fish that mature at later ages!.

Finally, spawning behavior of chinook salmon may to some extent alleviate the kind of long-term genetic shift toward younger age at maturity that might be expected to result from size-selective fisheries. Baxter (1991) found that larger and older chinook salmon, especially males, enjoyed greater reproductive success on spawning grounds that younger and smaller males. Thus, even if size-selective fisheries generated substantial shifts toward younger aged spawners, this kind of size-dependent mating success might at least partially buffer against such fishery-induced shifts to younger ages.

Ricker (1976) and Henry (1972) calculated the loss in potential yield that results from size-selective ocean fishery capture of immature and maturing chinook salmon as compared to terminal fishery capture of mature fish only. Calculated losses range from 30-50% of total yield. In two important reports, Ricker (1980, 1981) examined changes in average size of chinook salmon (and other Pacific salmon species) and presented a number of plausible hypotheses that might explain the apparent decline in average size of harvested chinook salmon. Included among these hypotheses was the possibility that size-selective fisheries had selected for long-term genetic changes in age at maturity. Hankin and Healey (1986) presented analysis of an age-structured Ricker stock-recruitment model and, among other things, attempted to calculate the maximum possible changes in mean age of spawning populations that could be explained as a direct consequence of *fishing-down* effects. They contrasted these calculated values with observed changes in mean ages in some populations. Hard (in press) used age-structured quantitative genetics models to assess the possible long-term genetic effects of size-selective fishing on chinook salmon populations

Stock-Selective Fisheries.

There seems little doubt that certain stock-selective fisheries must have long-term genetic effects on chinook salmon populations. Suppose, for example, that a terminal fishery were regulated by allowing harvest to take place only after a certain number of fish were estimated to have escaped to spawn. In that case, the fishery-related mortality rate would be much less for fish (or stock type) in the early part of the run than for fish (or stock type) in the late part of the run. Because run timing (stock type) is known to be an inherited trait, such fishery harvest policy should, in the long-term, unintentionally select for early-returning fish (or for a particular stock type). (See Nicholas and Hankin 1988 for examples of this phenomenon in a hatchery setting.)

Lawson and Sampson (1986) examined the potential impacts of stock-selective ocean fisheries on non-catch mortalities of species (e.g., coho vs chinook) or stock types (e.g., hatchery vs wild) that may not be landed in stock-selective fisheries. Such prohibited species or stock types would be captured but then released. Ricker (1958) presented modeling results showing that total yields in mixed stock ocean fisheries were considerably less than those that could be achieved if stocks could be managed and harvested separately. (This same theme was later noted by Hilborn (1985).
Evidence for Inheritance of Traits

Donaldson and Menasveta (1961) provide evidence that growth rate, survival rate, disease resistance and temperature tolerance are all traits which are subject to deliberate artificial selection in a hatchery setting. Ricker (1972) provides an extensive review of older studies that provide evidence that age at maturity and other traits are inherited trait, but also presents information on environmental influences on these same traits. By contrasting the rates of production of jacks in two chinook salmon stocks reared in a hatchery environment under controlled conditions, Hard et al. (1985) provide evidence that the tendency to produce age 2 male jacks is an inherited trait. Hankin et al. (1993) summarize evidence that age at maturity (all ages) is an inherited trait based on age-specific mating experiments carried out at Oregon's Elk River Hatchery. These analyses attempt to account for the fishery-induced biases that might result from differential mortality on older-maturing as compared to younger-maturing fish. Both Hankin (1993) and Hard et al. (1985) provide evidence that jacking rate does not depend on growth rate alone, but size nevertheless has an important effect (Hankin 1993, Silverstein et al. 1998), with faster-growing fish (at age) generally maturing earlier. If growth rates are sufficiently enhanced in hatchery environments, then mature yearling chinook can apparently be produced (Clark and Blackbird 1994). Heath et al. (1994a) carried out known matings designed to assess inheritance of jacking rate with male parents that were jacks or non-jacks. They found a significant sire age effect, but did not find that jacking was related to growth rate. Heath et al. (1994b) used DNA probes to show that allele distributions differed between maturing and immature chinook salmon of the same age and stock. Heath et al. (1999) presented experimental evidence for a maternal effect (via female egg size) on offspring size during early life (first several months, but thereafter no effect could be detected.

Behavior and Life History

Numerous papers have stressed the possible importance of large size in naturally spawning populations of chinook salmon. Baxter (1991) observed spawning behavior of fall chinook salmon in northern California and found that larger-sized males enjoyed much greater spawning success than smaller-sized males. Females exhibited behaviors suggesting their preference for mates that exceeded their size. Berejikian et al. (2000) found that there was a greater amount of time between successive nests for females paired with small males than with large males and suggested that this behavior might be an important means of achieving mate choice (i.e., finding a preferred larger-sized male. Healey and Heard (1984) examined variation in fecundity of chinook salmon among many chinook populations. Using life history models, they found that age-specific increases in fecundity would not "justify" the old ages at which many chinook salmon spawn. Presumably, there are some additional important benefits of large size and late age at maturation.

Egg size of chinook salmon varies across populations and within populations. Within a given population, egg sizes are generally larger for larger and older fish than for smaller and younger fish. Silver stein and Hershberger (1992) found that females with larger egg sizes were more likely to produce progeny that matured precociously. Healey (2001) reported that stream type chinook salmon, that typically spend more than a full year in freshwater prior to ocean entry, have smaller eggs and generally make a smaller reproductive investment than do ocean type chinook salmon, that typically enter saltwater during their first year of life.

Detecting Selective Effects of Fishing

Ricker (1980, 1981), previously mentioned, presented evidence for declines in average size and age of Pacific salmon, including chinook salmon, and listed a number of possible explanations for these declines. More recently, Bigler et al. (1996) found a decreasing average body size in 45 of 47 salmon populations in the Northern Pacific. They found that body size was inversely related to population abundance and speculated that enhancement programs during the 1980s and 1990s have increased population sizes but reduced growth rates due to competition for food in the ocean. Clearly, these kinds of causes could result in the same kinds of reductions in size at age as might be caused by long-term genetic selection against fast-growing fish.

There is substantial cause for concern regarding long-term genetic effects of both stock-selective and size-selective fishing on chinook salmon stocks. Of these two kinds of selective fisheries, the effects of stock-selective fisheries seem most clear and most easily minimized. If terminal fisheries consistently result in substantial removal of specific temporal components of a stock's spawning run, then it seems inevitable that there will be strong selection against perpetuation of these temporal components. This kind of effect would seem avoidable by regulating open and closed terminal fishing periods so that continuous fishing periods are always short (say, no more than 3 days duration), and so that the duration of fishing periods is always short compared to the duration of closed periods. Terminal net fisheries in Puget Sound are scheduled in this manner – pulsed openings scheduled over the duration of the run.

It seems clear that size-selective ocean fishing on immature chinook salmon can shift the age distribution of adult spawners toward smaller and younger fish. A long-term genetic shift to younger aged spawners would result (1) *If* chinook salmon mated randomly, without regard to age, on spawning grounds, and (2) *if* age at maturity were independent of growth rate. However, (3) larger and older male chinook salmon (and possibly females) generally have greater mating success than smaller and younger male chinook salmon (and possibly females); (4) fast-growing chinook salmon tend to mature at younger ages than slow-growing chinook salmon, but are selected *against* in size-selective ocean fisheries; and (5) size at age may have only a weak correlation with some inherent genetically inherited "growth rate". Together, items (3)-(5) may reverse or ameliorate the kinds of long-term genetic effects that one might expect if items (1) and (2) were valid. Most of these potential long-term genetic effects again seem avoidable. If ocean fishing for chinook salmon were prohibited by regulation (see Ricker 1976 for one example calculation of the improved yield that could result!), and if all sizes and ages of chinook salmon were equally vulnerable to terminal fisheries (e.g., by fishing gill nets of variable mesh sizes in Indian fisheries), then it would seem unlikely to expect any long-term genetic changes in age at maturity of chinook salmon stocks.

The absence of explicit consideration of possible long-term genetic impacts of selective fishing in management plans for chinook salmon stocks probably reflects the ambiguity and complexity of potential impacts for this species. No chinook salmon stocks have yet been reduced to the extreme scenario (only jacks and age 3 females) sketched by Ricker (1980, 1981), but it is also certainly true that one would be hard-pressed to find a stock of chinook salmon for which one might claim that the largest fish seen today are as large as those seen 100 years ago. Of course, given classical fishing-down effect that results from ocean fisheries, one would not expect to see these large fish even if there were no long-term genetic changes in age or size at maturity.